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HOW TO MAKE LAND SUBDIVISION WORK

AN ANALYSIS OF THE ECOLOGICAL AND SOCIO-ECONOMIC FACTORS AFFECTING CONSERVATION OUTCOMES DURING LAND PRIVATISATION IN KENYAN MAASAILAND

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ABSTRACT

The Amboseli-Tsavo Ecosystem, Kenya, represents one of the world's richest assemblages of mammalian fauna. The majority of the area is under no official protection, consisting of Maasai group ranches which are currently undergoing a transition from vast areas under communal ownership to small, individually-owned private land parcels. This land privatisation threatens the integrity of the ecosystem and, considering the conservation potential of the area, necessitates management from an early stage.

This multi-disciplinary thesis investigates the issue of land privatisation from an ecological perspective and then takes an economic angle to quantify the costs of living with wildlife as a Maasai pastoralist. A sociological component investigates local Maasai opinions towards wildlife and its conservation. Two neighbouring Maasai ranches were studied for comparative purposes; one (Merueshi) which was privatised and settled in the early 1980s and one which is only beginning to do so now (Mbirikani).

One major effect of land privatisation is pastoral sedentarisation and results have clearly indicated negative ecological consequences of these processes. On the privatised (and sedentarised) ranch, both grass quantity and local wildlife populations were significantly lower than on the communal one, with evidence presented to suggest this was not the case prior to land privatisation.

The privatised ranch had no tourist infrastructure and received no financial benefits from wildlife, while the communal one had a safari lodge and community conservation trust, with members benefiting financially. Attitudes towards wildlife and conservation were significantly more positive on the latter ranch, despite the considerable net cost to wildlife faced by households. All results highlight the importance of constructing a conservation plan for Mbirikani Group Ranch as it begins the land privatisation process, incorporating efforts to maintain communal use of the rangelands and to increase wildlife revenues and community participation in conservation.

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AUTHORS DECLARATION

I declare that the work in this dissertation was carried out in accordance with the regulations of the University of Bristol. Much of the field data was collected with the help of Maasai research assistants under my direction, and some of the data presented in Chapter 2 is from Dr David Western's long term study of the Amboseli Ecosystem. With this exception, this work is original and no part of the dissertation has been submitted for any other academic award. Any views expressed in the dissertation are those of the author.

SIGNED: ...Rosemary Groom.............. DATE: ....01/02/2008.......

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CHAPTER 1

INTRODUCTION TO THESIS
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ABSTRACT

This thesis takes a multi-disciplinary approach to investigating the issues of land subdivision and sedentarisation of the Kenyan Maasai. The aim was to gather sufficient data to suggest a management plan for the Maasai ranches within the Amboseli-Tsavo Ecosystem as they privatise previously communal rangelands, in what can be considered one of the world's most valuable wildlife areas. In order to understand the significance and importance of this study in a global context as well as locally, this chapter provides a review of the literature on pastoralists around the world. I investigated the changes seen in pastoral societies globally and the impact of such changes. I then briefly discuss the prognosis for the future of these pastoral societies, before discussing the situation in Kenya in more detail. Ultimately, I focus on the Maasai and the land subdivision within the Amboseli-Tsavo Ecosystem of southern Kenya. The chapter concludes by describing the rationale for the study, the overall aims and research questions of this thesis, and gives details of the specific study area in which the research was conducted.

1.1 BACKGROUND INFORMATION / LITERATURE REVIEW

1.1.1 Pastoralism worldwide

Pastoral production comprises about 25-33% of the world's land area and supports some 20-30 million pastoral households (Sanford 1976; Blench 2001). Pastoralism exists on every continent, with the exception of Antarctica (Blench 2001), and is one of the key production systems in the world's dry rangelands (Abule, Synman & Smit 2005). Pastoralism has been practiced worldwide for many centuries. For example, clearly identifiable domestic stock first appeared on the African continent after 7700 BP (Wendorf & Schild 1980 in Smith 1992).

Pastoralism is defined as a method of mobile livestock production that makes extensive use of grazing lands (Blench 2001; Markakis 2004). It can take several forms: a) Pure nomadism, where the people are exclusively pastoralists, do not grow crops and are permanently on the move following water and pastures. This is typical in Saudi Arabia and extremes of Sub-Saharan Africa. b) Transhumance, where people regularly move herds between fixed points to exploit seasonal availability of pastures. This can be a vertical movement, such as practiced by pastoralists in the Himalayas, the Alps, the Andes and in North Africa, or a horizontal movement which is common and widespread throughout Sub-Saharan Africa and West Africa and also found in India and the Syrian steppe (see Leybourne, Jaubert & Tutwiler 1993; Nautival *et al.* 2003; Abule *et al.* 2005). c) Agropastoralism, where people are settled and undertake subsistence cultivation as well as livestock farming. They tend to have smaller herds and depend on grazing within a day's walk of their settlement. This is common in East Africa (Graham 1988). In addition to these traditional pastoral systems, ranching, where the land is usually individually owned and fenced can also be considered a form of pastoralism. This is the dominant system in North America, Australia and parts of South America, notably Argentina and Brazil (Blench 2001).

1.1.2 Changes in pastoralism worldwide

Traditional, subsistence-oriented migratory pastoralism has declined markedly in much of the world's rangelands (Schwartz 2005). The major change, common to pastoralists throughout the world, is sedentarisation. Sedentarisation is defined as the settlement of previously nomadic or semi-nomadic people into permanent homesteads with a corresponding decrease in mobility of the people and their livestock (Salzman 1980). It is often accompanied by increasingly diverse livelihoods, in comparison with a near-exclusive reliance on livestock products, and can have both positive and negative consequences, depending on circumstance.

Causes (and consequences) of sedentarisation vary (Fratkin 1992), and the process can be both internally or externally driven. External drivers (usually 'push' factors) include public and government policies, weak land tenure and loss of land, population growth, increasing social insecurities and drought and famine (Njoka 1979; Salzman 1980; Roth & Fratkin 2005). Internally driven (voluntary) sedentarisation is characteristically driven by economic opportunities such as commercialisation and urban migration (Bates 1980; Chatty 1980), as well as a desire to be near schools and churches. Additionally, now that technology has improved, especially in the agricultural sector, more opportunities are available to pastoralists and sedentarisation is often necessary to take advantage of these and to capture the higher incomes available from non-pastoralist activities. Below I explore each of these in turn, and illustrate how and why, through a combination of historical policies and current events, traditional nomadic pastoralism has declined.

1.1.2.1 Public and government policies

Pastoralists usually belong to ethnic minorities and live in remote areas, and are therefore often only weakly represented at the national level (Leloup 2006). Historically therefore, public policies have rarely taken into account the needs of this group. For example, in Africa during the colonial era, centralized administrations restricted pastoral movements within administrative units, irrespective of the ecological needs of the pastoralists (Leloup 2006). Since then, resource tenure systems were established that neglected existing customary tenure systems and undermined relevant local authorities, especially with regard to the use of natural rangelands (Kirk 2000 in Leloup 2006).

Specifically, national governments in Africa and elsewhere have long been concerned with sedentarising pastoralists (Blench 2001; Roth & Fratkin 2005). Indeed it has been implied that the aspiration of a modern state seems to be to erode the value system of a nomad and replace it with the value system of the state (Njoka 1979). For example, after the revolution of 1917, Soviet Russia embarked on an extensive sedentarisation program among both the Kazakhstan and Kirghizia tribes, in order to "fuse the tribal loyalties into higher loyalties of the state" (UNDP 1967 in Njoka 1979). In addition, nomadic groups in Iran and Israel have settled in response to state enforced measures (Roth & Fratkin 2005).

Governments worldwide promote settlement with the aim of intensifying and commercialising livestock production, as they seek to provide cheaper meat to urban areas (Pratt, Le Gall & de Haan 1997), and expect the pastoralists to make a greater contribution to the national economy (Dyson-Hudson & Dyson-Hudson 1982; Graham 1988; Homewood & Rodgers 1991; Cochrane *et al.* 2005). For example, in the last few decades, traditional pastoralism in Saudi Arabia has been transformed into highly mechanized, high-input grazing systems (Ahmad 2001 in Schwartz 2005). In many countries, this market integration has forced a shift from subsistence pastoralism to specialization, such as meat production in Algeria or camel milk production in parts of Syria (Schwartz 2005).

Biased incentive policies at both the national and international level have also disfavoured pastoralists. For example, in many parts of Africa, government policies to attain food security have favoured cropping systems for cereals over pastoral systems through subsidising of agricultural equipment and input prices (Pratt *et al.* 1997). Moreover, subsidies of commercial livestock ranching, at the expense of rangelands for pastoralists and wildlife, creates further policy distortions (Cullis & Watson 2004 in Leloup 2006). In addition, the international dumping of beef by the EU under its Common Agricultural Policy (which was favoured by African governments because it provided cheap meat to urban areas) reduced the income of pastoralists. This was especially the case in West Africa, where it caused pastoralists to give up their mobile lifestyle and take up arable farming to support their subsistence and monetary needs (Leloup 2006).

Involuntary settlement of pastoralists has also been reported because of dam construction, famine and civil war (Larsen & Hassan 2003). For example, Somalia has experienced spontaneous, sometimes violent, privatization of communal rangelands for sedentary livestock production (Schwartz 1993 in Schwartz 2005).

1.1.2.2 Weak land tenure and loss of land

Increasing pressure on pastoral lands is occurring worldwide and the usually weak tenure of such lands is often a problem. For example, in the Middle East and Iran, governments have declared semi-arid and/or non-cultivated land to be state property, with grazing rights for pastoralists either withdrawn or tightly controlled (see Galaty 1980). In Africa (and elsewhere) since the early 1900s, vast areas of natural rangelands have been taken over for agriculture, private livestock and/or game ranches, nature reserves and infrastructure (see Galaty 1980; Cochrane *et al.* 2005; Leloup 2006), and in Syria, the Bedouin lost much of their pastoral rangelands to agriculture (Leybourne *et al.* 1993). This land encroachment occurs most frequently in the best dry season grazing areas, which are the key resources ensuring overall sustainability of the pastoral system (Box 1971; Leloup 2006). This not only reduces traditional long distance migrations to much shorter grazing treks, but has engendered a fear of further land dispossession which creates social problems and encourages settlement and land fragmentation through a desire for official title deeds. Moreover, this insecurity of tenure may prevent the pastoralists from investing in good, sustainable land use practices. This is occurring throughout the world, with good examples provided by the Maasai in Kenya (Galaty 1980), the Aboriginals in Australia (Gill 2005) and the Oromo in Ethiopia (Abule *et al.* 2005).

1.1.2.3 Population growth

Population growth is occurring at various rates throughout the world and is often high in pastoral areas. An increasing number of pastoralists inevitably leads to greater pressure on the land and an ultimate reduction in the ability of the rangelands to support both livestock and people (Talbot 1986). Not everyone can remain nomadic pastoralists once the carrying capacity of the land has been met. For example, it has been suggested that the rangelands of Sub-Saharan Africa are already at their maximum potential and the outlook for supporting a growing population is bleak (Breman & de Wit 1983). Moreover, increasing populations of agriculturalists who live adjacent to the rangelands often causes further encroachment into the wetter areas of the rangelands, reducing the grazing land available to the pastoralists (Talbot 1986). Effectively, increasing human populations intensify the demand for the rangeland resources beyond the ability of the land to provide them (Talbot 1986), often resulting in serious environmental effects and forcing people to settle.

1.1.2.4 Increasing social insecurities

It is often the case that as pressure on resources intensifies, so does conflict over the same resource, and enduring political tensions may result from land encroachment (Leloup 2006). In some parts of the world, political turmoil and violence have reduced pastoralists' accessibility to their land and influenced pastoral sedentarisation. For example, northern Kenya has been fraught with insecurities since the early twentieth century. In the early-mid 1900s, the Gabra were consistently attacked by the Dassanach and Ethiopians (Galaty 2005). Clashes continued between the Gabra and the Dassanach, as well as between the Dassanach and the Rendille in Marsabit District (Galaty 2005). Such conflicts resulted in a loss of capacity for some members of this community to continue pastoralism, and as such, the patterns of sedentarisation are defined by the patterns of conflict (Galaty 2005). In addition, a drawn-out strife between the Rendille and Gabra, beginning in the early 1900s, has also influenced settlement. For example in 1992, the Gabra became stronger in terms of firearms, which forced a southward retreat of the Rendille, accelerating the process of sedentarisation as the Rendille gathered in larger and larger settlements (Galaty 2005).

This illustrates how the terrorization of local communities by periodic insecurity has increased pastoralist inertia, resulting in increased stasis of the herds as extensive grazing becomes less secure (Galaty 2005). All these factors lead to an increase in pastoral sedentarisation.

1.1.2.5 Drought and famine

The recurrence of drought is sometimes considered one of the most important forces leading to massive and sudden sedentarisation of pastoralists in Africa (Njoka 1979). This can be either voluntary settlement, or state-controlled movement. An example of the latter is provided by the Somali pastoralists, of whom 168,000 were moved from drought stricken areas after the 1973/74 drought, into settlement areas 700 miles away (Njoka 1979). In the Syrian steppe, a three-year drought from 1958 to 1961 dramatically reduced the camel population, a partial consequence of which was the decline in nomadic lifestyles of the Syrian Bedouin (Leybourne *et al.* 1993). In addition, the provision of relief food aid in drought times can encourage settlement. For example, famine relief efforts by the Catholic Church in the Marsabit District of Kenya after the extensive droughts of the 1970s and 1980s contributed to the settling of former nomads (Fratkin 1992). These aid efforts led to the growth of small towns, and further projects such as UNESCO's Integrated Project in Arid Lands focussed on the improvement of livestock marketing, which also increased sedentarisation (Fratkin 1992).

1.1.2.6 Improved economic opportunities and access to infrastructure

Despite the negative forces described above, for many pastoralists, settlement is not obligated either by government policy or circumstance, but is rather a voluntary change of lifestyle. Towns and villages offer opportunities to sell livestock and agricultural products. They are close to schools and health care and have prospects for wage-paying employment (Roth & Fratkin 2005). Indeed, pastoral sedentarisation in East Africa in the 19th and early 20th centuries was prompted largely by new market opportunities (commercialisation) rather than the population pressure and ecological decline that has characterised the 20th and early 21st centuries (Roth & Fratkin 2005). Access to permanent water can also be a strong influence in increasing voluntary sedentarisation. Additionally, engaging in agriculture is considerably more profitable than pastoralism (Norton-Griffiths *et al.* in press), and where rainfall allows, pastoral settlement to undertake agriculture is becoming increasingly common. Whilst this can be considered disadvantageous for remaining pastoralists who lose land to such activities, it usually substantially increases the wealth and opportunities of those who do it.

Furthermore, in recent decades, people's expectations have changed. The younger generation of pastoralists are seeking more comfortable standards of living, the achievement of which necessitates a more settled existence in order to access infrastructure and social services such as education, health care and veterinary care (Pratt *et al.* 1997). Moreover, the attainment of property rights through privatisation of communal lands gives pastoralists individual wealth which can be used for investment, or the security for applying for loans, both of which allow development and a considerable improvement in lifestyle. Today, wealth from pastoralism itself can increase sedentarisation through the ability to transport water to livestock using trucks rather than having to move livestock to water (pers. obs.).

In conclusion, the process of sedentarisation is undertaken in a variety of ways and for a variety of different reasons. It may be a voluntary process, undertaken in order to maximise economic or political opportunities, or it may be externally driven, either through government policy, pressure on the land or through drought and famine. Consequences of sedentarisation also vary, although there are some general trends which appear similar worldwide. I discuss these general impacts below, then later more specifically for the Maasai of southern Kenya.

1.1.3 The impact of these changes

Sedentarisation is not only a recent event (Roth & Fratkin 2005). Bulliet (1980) describes the sedentarisation of Arabic nomads in southern Iraq in the seventh century. In addition, the Oromo people of Kenya were settled and participating in the market economy for most of the 20th century (Ensminger 1992) and Swidler (1980) discusses the sedentarisation of several different pastoral groups in the Middle East since the early 1900s. There has therefore been ample opportunity to study the impacts of sedentarisation.

Recent studies on pastoral sedentarisation have described a variety of costs and benefits to such a change in lifestyle. Several studies point to problems of impoverishment for pastoralists who settle (Little 1985; Talle 1999), whilst others point out increased marketing benefits (Ensminger 1992; Zaal & Dietz 1999), or the benefits of switching to agriculture

(Norton-Griffiths *et al.* in press). Sedentarisation has important implications for health and nutrition, and tends to change the traditional social structures and customs of pastoral societies [for a review from northern Kenya, see Fratkin & Roth (2005)]. Sedentarisation of previously nomadic people can also have major consequences for the environment, and it is these impacts which are the focus of this study.

Many authors report that declining mobility of pastoralists leads to environmental degradation and increased poverty (Darling & Farver 1972; Talle 1999). Indeed, environmental degradation is one of the most frequently cited consequences of sedentarisation (Salzman 1980; Roth & Fratkin 2005). Overgrazing and land degradation occur to a greater extent when livestock is forced to stay in a restricted area (Boone 2005; Leloup 2006) and constant grazing pressure in a contained area reduces the root stock available and can result in soil erosion as well (Kimani & Pickard 1998). A decline in the *quality* of vegetation present is also a recognised feature of settlement and consequent heavy grazing pressure (Boone 2005). On the contrary, land degradation from *mobile* pastoralism is often temporary, and the resilient vegetation tends to restore itself if given a season without grazing. Additionally, it is well established that intermediate and rotational grazing pressure can increase the quality of the grasslands (e.g. Guevara, Stasi & Estevez 1996), but this positive effect is lost when pastoralists settle.

Other impacts of sedentarisation include changes in herd ownership and declining herd production (Leloup 2006). *Per capita* ownership of livestock is declining significantly (see Leybourne *et al.* 1993), owing in part to the increasing human population, and in part to the inability to maintain such large herds without extensive freedom of movement. For many pastoralists, *per capita* head of livestock is now below the minimum subsistence level (Leloup 2006). This is not necessarily a bad thing however, and does not indicate increasing poverty unless the household is entirely dependent on livestock. In many cases, a decrease in livestock holdings *per capita* is accompanied by a diversification of livelihoods which can make households considerably wealthier. Nonetheless, standards of living *are* reportedly falling amongst the mobile pastoralists of Africa (Talle 1999; Leloup 2006), often resulting in further settlement due to the need for other sources of income such as crop farming, wage-labour or food aid (Niamir-Fuller 1999). In many parts of Africa, frequent or even permanent food aid or other technical interventions are an attempt to alleviate this poverty (Pratt *et al.* 1997; Hazzah 2007). This in turn has its own consequences on pastoral livestock production systems, in some cases reducing dependency on livestock for subsistence, such that husbandry becomes careless and conflict with wild carnivores increases (see Hazzah 2007).

1.1.3.1 The effects on wildlife

Of particular relevance to this thesis is how these changes may affect wildlife populations. Pastoralists in the past have often co-existed fairly peaceably with wildlife (Akama 1999; Campbell *et al.* 2000; Seno & Shaw 2002). However, increasing pressure on the land, through both population growth and sedentarisation, and the increased emphasis on a monetary economy (McCabe 2003) have resulted in decreasing tolerance of and increasing conflict with wildlife (Norton-Griffiths 1995; Thompson & Homewood 2002). Wildlife declines resulting from sedentarisation among the Kenyan Maasai are discussed below (Section 1.1.7.3).

1.1.3.2 Conclusion

Examples presented above have illustrated how the process of sedentarisation is a complex one, with varying consequences depending on region, government and opportunities for change. In general, the sedentarisation of pastoralists has contributed to economic differentiation and a shift towards wage-based labour in many rural areas (Fratkin 1992). Many local economies are now based on a combination of subsistence pastoralism, wage-labour and livestock marketing (Fratkin 1992). Whilst sedentarisation may have negative ecological consequences, in some cases it may be a positive development for the individual or household which chooses to settle.

1.1.4 The future of mobile pastoralism

For reasons discussed, mobile pastoralism is on the decline throughout the world. Nonetheless, despite a global trend towards settlement (Robbins 1998), in some places nomadic pastoralism is becoming popular once again. For example, the incidence of pastoral nomadism is on the rise in the Marwar region of Rajasthan, India (Robbins 1998) and the San hunters of South Africa are shifting to herding (Smith 1990 in Smith 1992). Additionally, in 1990 Mongolian livestock farmers began to move away from the state-controlled, centralised livestock production system to more traditional rangeland management practices (Rasmussen *et al.* 1999).

There is also a growing recognition of the fact that, despite previous schools of thought largely resulting from inadequate and poorly focussed research (Leloup 2006), mobile pastoralism is not a 'waste of space' or an irrational and inefficient use of the land (Herskovits 1926; Brown 1971), but is rather a highly specialized livestock production system which utilises the natural resources of marginal areas in an efficient and sustainable fashion (Breman & de Wit 1983; Ellis & Swift 1988; Hesse & MacGregor 2006). Moreover, many of the negative impacts of sedentarisation that I discussed above are beginning to be recognised by pastoralists, governments and scientists alike, and more effort is being put into maintaining more flexible, mobile systems (Niamir-Fuller 1999; Ostrom *et al.* 1999).

Nonetheless, for the most part, pure pastoral nomadism is a thing of the past and pastoralists in the future will live different lifestyles and face different challenges during their more sedentary existence. Hand-fed supplementary feeding, now relied on intensively by the Syrian Bedouin (Leybourne *et al.* 1993), may become an essential component of livestock rearing in the future, as the availability of natural rangeland resources decreases. Intensification of farming practices is inevitable to some degree, and increasing mechanization of livestock production may begin to occur. It is likely that where rainfall is sufficient, increasing crop farming will take place (Leybourne *et al.* 1993), combined with an increased dependence on agricultural products in the diet. Rangeland degradation resulting from settlements as well as the increasing pressure on the land from human population increase is likely to make it increasingly difficult to return to a nomadic lifestyle, despite the fact that in the past, sedentarisation has been considered reversible (Salzman 1980).

The issues of changing pastoral lifestyles and their causes and consequences are now discussed in detail for the Maasai in Kenya.

1.1.5 Kenya - its people and wildlife

Kenya's population is about 30 million people (Ottichilo *et al.* 2000; Roth & Fratkin 2005). Of these, around 2 million are considered pastoralists (Roth & Fratkin 2005), with a further 2-3 million agro-pastoralists (Norton-Griffiths 1998; Markakis 2004). The arid and semi-arid lands of Kenya comprise 80-90% of the country's total land surface (541,416km^2) (Ng'ethe 1993; Ottichilo *et al.* 2000). They support about 35% of the human population and over 50% of the country's livestock (Government of Kenya 1992; Ng'ethe 1993). Most of these rangelands are under nomadic pastoralism, with the pastoralists inhabiting 70% of Kenya's land (Ottichilo *et al.* 2000; Roth & Fratkin 2005). Kenya's pastoral tribes include Nilotic speaking groups such as Maasai, Samburu, Pokot, Turkana and Chamus, and Afro-Asiatic speakers including Boran, Gabra, Rendille, Sakuye and Somali (Dyson-Hudson & Dyson-Hudson 1982; McPeak 2005; Roth & Fratkin 2005).

As well as its cultural diversity, Kenya has an enormously diverse wildlife population. In fact East African savannahs are famous throughout the world for supporting the planet's richest variety and density of large mammals (Little 1996; Du Toit & Cumming 1999). By the end of the 20th century, Kenya had 26 national parks, 26 national reserves and several nature reserves and animal sanctuaries. These occupy around 7-8% of Kenya's total land area (Kinyua, van Kooten & Bulte 2000; Ottichilo *et al.* 2000; Markakis 2004). Nonetheless, at the last estimate, over 70% of Kenya's wildlife was found outside of parks and reserves (Grunblatt *et al.* 1995a; Norton-Griffiths 1998). Although it is probable that this number is lower now, it is still clear that a significant proportion of Kenya's wildlife depends on the pastoral rangelands. Since parks and reserves are generally accepted as being too small to conserve the current abundance and diversity of wildlife found in Kenya (Western & Gichohi 1993), a great responsibility is placed on Kenya's pastoralists for the conservation of wildlife on their lands.

However, since 1977, populations of all wildlife species in Kenya except wildebeest and ostrich have declined significantly (Ottichilo *et al.* 2000). For example, the national wild herbivore populations in the rangelands showed a decrease of 40-60% between the 1970s and the 1990s (Grunblatt, Said & Wargute 1996). The main factors contributing to this decline include poaching and land use change (Ottichilo *et al.* 2000), and the hunting ban in 1977 which removed the opportunity for many communities to benefit economically from their wildlife (Norton-Griffiths 2007).

Many of the country's parks and reserves (e.g. Amboseli National Park, Tsavo West National Park and the Maasai-Mara National Reserve) lie within former Maasai territory and are currently surrounded by Maasai group ranches. Much of the wildlife outside of protected areas is also found on Maasai lands. Consequently the Maasai people are instrumental in the conservation of Kenya's wildlife assets, and as they are also undergoing a process of sedentarisation, along with other land use changes, they provide an important opportunity to study the impacts of sedentarisation on the people, wildlife and ecosystem.

1.1.6 The Maasai

Both historians and anthropologists describe the Maasai as one of the most prominent and powerful communities in East Africa prior to the mid-19th century (Hazzah 2007). The Maa-speaking pastoral Maasai inhabit the arid and semi-arid grazing lands in eastern Africa. Today, the Maasai lands stretch from the Kenyan Loita-Mara plains, across the Serengeti to the Ngorongoro Crater, and toward the southern plains of Tanzania (Ojalammi 2006). Evangelou (1984) estimated the total Maasai population to be around 280,000 (180,000 in Kenya, and 100,000 in Tanzania). More recent estimates put the total population at 350,000 in 1997 (Fratkin 1997) and 750,000 in 2006; 400,000 in Kenya and 350,000 in Tanzania (Ojalammi 2006).

The Maasai are traditionally semi-nomadic pastoralists and represent an extreme case of pastoral dependency on livestock. They are viewed by many as 'people of cattle' (Spear & Waller 1993; Anderson 1995), whose livelihoods depend on access to vast stretches of pasture and widely distributed water sources (Hazzah 2007). However, pure Maasai pastoralism has declined during the twentieth century (Spear & Waller 1993), owing to both constraints on their nomadic lifestyles and improving economic opportunities *off* the land. While some households do remain entirely dependent on livestock, with a minimal reliance on non-pastoral produce in their diets and daily lives (Galaty 1980), many are now agro-pastoralists (Homewood *et al.* 2001) or wage-earning employees in cities and towns. For example, over the last three to four decades, many Maasai have barely survived on their livestock holdings and have been forced to seek income from other sources (Kituyi 1990). This usually leads to Maasai men moving away from pastoral areas into towns where they often find employment as night watchmen, whilst the women turn to petty trading, beer brewing and increasingly to prostitution (Talle 1999).

1.1.7 Maasailand - the changes

1.1.7.1 Historical context

The Maasai have dominated the pastoral niche in East Africa for the past four centuries (Spear & Waller 1993). Prior to European colonization in the late 1800s, Maasai inhabited well over 200,000 km^2 of land in Kenya and present-day Tanzania (Talbot 1986). However, a host of misfortunes including inter-tribal warfare, disease and drought weakened the Maasai power and made them vulnerable to British rule (Hazzah 2007).

Through the Anglo-Masai Treaties of 1904 and 1911, the British dispossessed the Maasai of much of the fertile Rift Valley (Halderman 1987), and relocated them to prescribed reserves (Sindiga 1984; Hazzah 2007). By 1913, the area of land occupied by Maasai had been reduced to 40,000km^2 (Grandin 1991).

This reduction in land, coupled with inevitable population growth, meant that the Maasai experienced a 5.5 fold increase in population density in less than 30 years (Talbot 1986). Livestock numbers increased correspondingly, until pressure on the land became so high that land degradation began to occur (Hazzah 2007). However, British attempts to reduce livestock numbers were inconceivable to the Maasai because this was analogous to stripping away their pastoral identity (Spear 1993). In addition, it would increase their vulnerability to environmental risks (Hazzah 2007).

Subsequent to the surrender of land to colonial settlers, the Maasai continued to lose land to other tribes and to the protection of wildlife (Sindiga 1984; Kimani & Pickard 1998; Akama 1999; Campbell *et al.* 2000). For example, in 1945 the government began gazetting a series of national parks and reserves at the request of European hunters and conservationists (Hazzah 2007), many of which where located in former Maasai territory. This further loss of land created a fear of dispossession and can help to explain the insecurity of tenure experienced by many of today's Maasai.

Currently in Kenya, there are two main Maasai districts, Kajiado and Narok. In these districts, Maasai numbers have increased substantially over the last few decades, although the proportion of Maasai in the total population has decreased noticeably (Rutten 1992; Coast 2002) due to immigration by other tribes (Ntiati 2002). For example, human population in the Kajiado District rose from 85,903 people in 1969 to 258,659 in 1989 (Republic of Kenya 1994) to 405,000 in 1999 (Campbell *et al.* 2003) and was predicted to be 502,861 in 2001 (Republic of Kenya 1997). The annual population growth rate of the district is approximately 5.54% per annum (Republic of Kenya 1997).

1.1.7.2 Group ranches

This increasing pressure on the land and a fear of dispossession amongst the Maasai led the Government of Kenya to propose the group ranch concept in an attempt to transform the nomadic subsistence production system into a commercially organised, more sedentary system (Graham 1988; Grandin 1991; Seno & Shaw 2002). Group ranches were officially initiated in 1968 by the Land (Group Representatives) Act (Grandin 1991; Fratkin 1997; Ntiati 2002), which gave each member a freehold title deed to undivided shares in the group ranch (Kimani & Pickard 1998; Campbell *et al.* 2003). There were originally 159 group ranches in Kenya, of which 51 were in the Kajiado District (Ng'ethe 1993). The main aims of the group ranches were to reduce stocking rates in pastoral areas, increase the Maasai's contribution to the national economy and prevent landlessness among pastoralists (Galaty 1980; Kimani & Pickard 1998).

A group ranch is officially defined as a livestock production system or enterprise where a group of people jointly own freehold title to land, maintain agreed stocking levels and communally herd their individually-owned livestock (Ministry of Agriculture 1968). Selection of members was based on kinship and traditional land rights (Ng'ethe 1993). One of the main aims of group ranches was to address the problems of overgrazing and land degradation by encouraging the Maasai to reduce their herd sizes and confine them within ranch boundaries (Kimani & Pickard 1998). However, most Maasai did not seem to fully understand the consequences or expectations of group ranches (Bekure & de Leeuw 1991; Rutten 1992). Nonetheless, they accepted them because they provided some security of tenure and ensured them exclusive rights to grazing land (Graham 1988; Grandin 1991; Kimani & Pickard 1998). However, rather than decreasing herd sizes and reducing mobility, most Maasai continued using their land along traditional lines, negating the group ranch concept (Campbell 1984; Rutten 1992).

In fact, the group ranch concept was flawed from the start, as many of them failed to include dry and wet season pastures within their boundaries (Halderman 1987; Graham 1988), which had been the original aim. In the Kaputei region of Kajiado District, the group ranches demarcated were so small it was doubtful they would be able to operate autonomously without occasional use of common resources (Galaty 1980). In addition, inefficient and corrupt management by group ranch committees, pressure to register the next generation as ranch members and continued insecurity of land tenure (Ng'ethe 1993; Kimani & Pickard 1998) led to a call for group ranch subdivision and land privatisation (Grandin 1991).

1.1.7.3 Land privatisation and subdivision

In Kenya, the policy of land privatisation was accepted and encouraged by the government from 1983 (Norton-Griffiths 1998). It had in fact been a provision of the original Group

Representative Act in 1968 (Ng'ethe 1993). Land privatisation in this context refers to the legal division of communally owned group ranch land into individual family-owned smallholdings. Whilst the term land subdivision is often used to refer to this process, in this thesis, I use a more literal definition of land subdivision, i.e. the physical division of the land by fencing of land parcels or simply by an enforcement of one's private property rights. This is an important distinction, since private land ownership does not necessarily entail a division of the rangelands if there is continued cooperation between land owners and communal grazing associations are maintained.

Land privatisation has been prompted by the failure of the group ranch system to deliver the pastoralists objectives of improved livelihoods and security of tenure (Ntiati 2002). From the government's point of view, group ranches also failed in their goal of range conservation and increased livestock off-take (Ng'ethe 1993). Norton-Griffiths *et al.* (in press) present three main drivers of land privatisation; 1) insecurity of tenure from political elites, conservation organisations and in-migration from other tribes, 2) a dilution of the value of communal resources following population growth and in-migration, and 3) a desire to capture the benefits of agricultural, livestock or wildlife production at the household level rather than through local institutions. The latter appears to be the main driver of privatisation and subdivision in the Mara, Kitengela and Machakos regions of Kenya (Norton-Griffiths *et al.* in press), largely due to corruption at the institutional level (Thompson & Homewood 2002).

In Kenya's Kajiado District, group ranch privatisation began in the 1980s with government support (Kimani & Pickard 1998). By 2006, 22 group ranches in Kajiado District (out of a total of 52) had been completely demarcated, and a further 15 were in the process (BurnSilver & Mwangi 2007). In general, the procedures used in the demarcation of the group ranches are characterised by a lack of a defined process and tend to be fairly *ad hoc* in nature (Ntiati 2002). In theory, land should be allocated by secret ballot, and the size of the parcel should be determined by the location and type of land, e.g. grazing land, irrigated land, areas suitable for rain-fed agriculture and conservation areas. In practice, local elites and families with influence often secure the best plots at the expense of others (Thompson & Homewood 2002).

The consequences of land privatisation and further subdivision are many and varied and include economic, social, cultural and ecological implications; both positive and negative. These consequences have been reviewed by Ng'ethe (1993), Kimani & Pickard (1998) and Ntiati (2002). They typically include sale of plots to non-Maasai, increased cultivation, an increase in the number of fenced plots, a decrease of livestock herd mobility leading to overgrazing and potential land degradation, increasing conflict between humans and wildlife and a decreasing tolerance of wildlife. Many studies report negative consequences of land subdivision for local wildlife populations (Seno & Shaw 2002; Worden, Reid & Gichohi 2003), especially when the plot sizes are small (Norton-Griffiths 1998). For example, Norton-Griffiths (1998) shows how every 1% decrease in the size of land holdings leads to a 2% loss of wildlife density and a 0.4% loss of wildlife diversity. This is partly due to interference with traditional wildlife migration patterns through fencing (Kimani & Pickard 1998).

Many of these radical implications come about because of loss of access to key resources by people, livestock and wildlife, and the increasing constraints on movement between remaining resources (Rutten 1992). In some cases, individual holdings may be too small to provide adequate family subsistence (Ng'ethe 1993), and such a scenario frequently leads to a permanent decline in mobility of the people and their livestock.

1.1.7.4 Sedentarisation of the Maasai in southern Kenya

In many instances, land privatisation leads to an increase in permanency of settlement due to the reduced flexibility of movement once land is privately owned (Cochrane *et al.* 2005). This sedentarisation represents one of the greatest socio-cultural transformations of pastoral nomadism (Njoka 1979). Government policy has played an important part in the sedentarisation of Kenya's nomadic tribes (see Markakis 2004), especially the Maasai. Indeed the main objective of Kenya's Arid and Semi-Arid Land (ASAL) Development Policy is "to improve the standard of living of the ASAL population by integrating ASAL into the mainstream of the national economy" (Government of Kenya 1992), which, as mentioned, is one of the major drivers of sedentarisation.

Sedentarisation of the Maasai has limited the two main methods of adaptation employed by these pastoralists: high mobility and stock-splitting, which enabled optimal use of marginal areas (Ng'ethe 1993). In this way it has made families increasingly vulnerable to the effects of major droughts (Cochrane *et al.* 2005) and has increased the need for livelihood diversification. Additionally, in the Maasai lands of southern Kenya, sedentarisation has occurred on the best dry season pastures, transforming these into agricultural lands and thus excluding them for pastoral use (Njoka 1979; Ntiati 2002).

Legal ownership of land, combined with sedentarisation, has led to more intensive resource use and land degradation where cultivation is attempted in marginal rain-fed areas (Njoka 1979). The process has also been partially responsible for the reduction in wildlife numbers in most subdivided areas of southern Kenya's Maasai lands (this thesis).

However, sedentarisation has also had some advantages. Where families are settled, it is easier to extend to them the important aspects of modern infrastructure such as education and healthcare, and schools and clinics are evident in the settled areas of southern Kenya's Maasai lands. In addition, with a settled population, it becomes easier to develop a market system and therefore integrate people into a monetary economy (Njoka 1979; Cochrane *et al.* 2005). Educational opportunities may eventually lead to a diversification of livelihoods (Cochrane *et al.* 2005).

1.1.7.5 Summary

In summary, social changes are taking place among the pastoral Maasai of southern Kenya, as they adjust to a dynamic, modern society. These changes are typical of changes facing pastoralists around the world, and in areas of conservation interest, are changes which need carefully managing for the benefit of the people and wildlife resource alike. This thesis focuses on exploring some of the ecological realities behind the process of land subdivision and sedentarisation, and investigates social and economic factors which are likely to influence pastoralists' decisions regarding wildlife conservation on their private lands. Both are investigated with the ultimate aim of producing a comprehensive management plan for the Maasai ranches of the Amboseli-Tsavo Ecosystem.

1.2 THESIS OBJECTIVES

To this extent, this study aims to address the following specific objectives, each of which is dealt with in detail in the following six chapters.

Data chapters

- To determine the environmental effects of land subdivision and increasing sedentarisation of nomadic pastoralists.

- To investigate the effects of land subdivision on the distribution patterns of wild grazers.

- To quantify the costs and benefits of wildlife to Maasai pastoralists in the Amboseli-Tsavo Ecosystem.

- To investigate the effect of wildlife revenues on the attitudes and behaviour of Maasai pastoralists in the Amboseli-Tsavo Ecosystem.

Discussion chapters

- To summarise the results of the four data chapters and discuss their implications as part of a holistic approach towards conservation planning.

- To present a detailed conservation plan for Mbirikani Group Ranch based on results of the four data chapters and the discussion of conservation planning literature in Chapter 6.

1.3 THE STUDY AREA

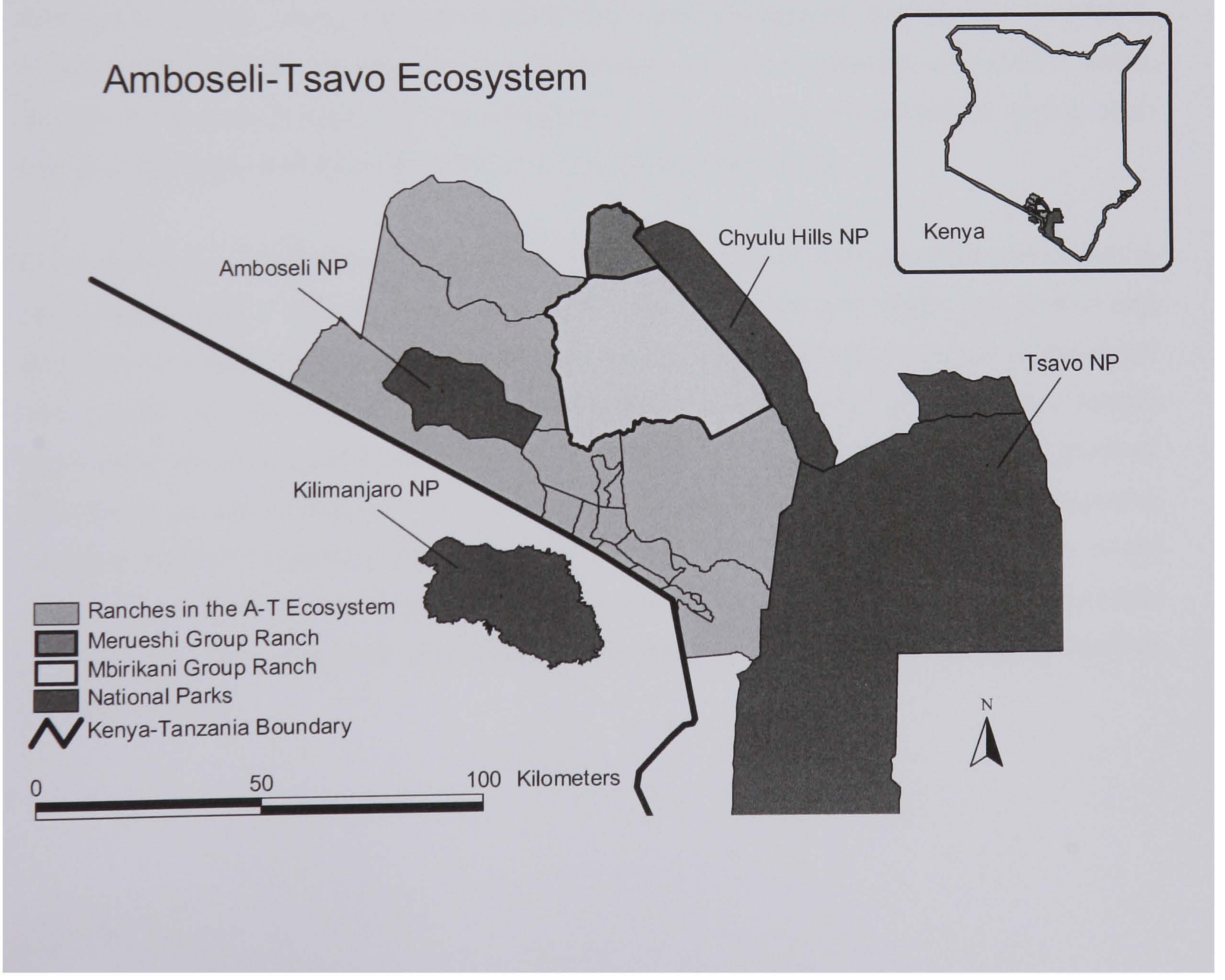
The Maasai lands of the Amboseli-Tsavo Ecosystem provide an ideal area in which to study the issues of land subdivision and sedentarisation of semi-nomadic pastoralists. The Kajiado District, of which the Amboseli-Tsavo Ecosystem is a part, comprises Maasai group ranches at all stages of the land subdivision process. The ecosystem also supports an abundance and diversity of wildlife and is one of the few places in East Africa where significant numbers of large mammals roam freely outside protected areas. It is also home to a rapidly growing human population which primarily exploits natural resources for subsistence and economic development. This has the potential to result in considerable conflict between farmers and the local wildlife and is therefore an important area to focus conservation efforts.

The Amboseli-Tsavo Ecosystem consists of six Maasai group ranches with a combined area of 5583km^2, and Amboseli National Park (392 km^2). The area supports approximately 36,000 Maasai pastoralists (Croze, Sayialel & Sitonik 2006), 410,000 head of livestock (Western & Manzolillo-Nightingale 2005) and an estimated 70,000 head of wildlife (this study). Key herbivore species include black rhino (*Diceros bicornis)*, elephant (*Loxodonta africana)*, buffalo (*Syncerus caffer)*, plains zebra (*Equus burchelli burchelli*), white-bearded wildebeest (*Connochaetes turinus albojubatus)*, Coke's hartebeest (kongoni) (*Alcelaphus buselaphus cokii)*, eland (*Taurotragus oryx)*, lesser kudu

(*Tragelaphus imberbis australis*), fringe-eared oryx (*Oryx gazella callotis*), gerenuk (*Litocranius walleri*), Maasai giraffe (*Giraffa camelopardalis tippelskirchi*), impala (*Aepyceros melampus*), Grant's gazelle (*Gazella granti*) and Thomson's gazelle (*Gazella thomsoni*). Carnivores found in the area include lion (*Panthera leo*), leopard (*Panthera pardus*), cheetah (*Acinonyx jubatus*), spotted hyaena (*Crocuta crocuta*), striped hyaena (*Hyaena hyaena*), wild dog (*Lycaon pictus*), black-backed jackal (*Canis mesomelas*), side-striped jackal (*Canis adustus*), bat-eared fox (*Otocyon megalotis*), serval (*Leptailurus serval*), caracal (*Caracal caracal*), civet (*Civettictus civetta*) and genet (*Genetta genetta*).

The specific study area includes one of these ranches, Mbirikani Group Ranch, and an adjoining ranch called Merueshi which is just outside what is officially considered the Amboseli-Tsavo Ecosystem (Figure 1.1). Land on both ranches is classified as Agroclimatic Zones V and VI (Bekure *et al.* 1991), arid to semi-arid.

Figure 1.1 A map showing the location of the two ranches studied within the Amboseli-Tsavo Ecosystem. Inset shows the location of the study area in Kenya.



Temperatures range from highs of 35^0C in February to lows of 8^0C in July (Altmann *et al.* 2002). Droughts are frequent and during the past century droughts have been recorded in 1933-35, 1943-46, 1948-49, 1952-53, 1960-61, 1972-76, 1983-84 and 1994-95 (Campbell 1999; Roth & Fratkin 2005). So far this century, 2000-01 and 2005-06 have been drought years (Roth & Fratkin 2005).

1.3.1 Mbirikani Group Ranch

Mbirikani Group Ranch was approximately 321,100 acres, bordered on the eastern edge by the Chyulu Hills National Park, which connects it to Tsavo West National Park, and with Amboseli National Park close to the western boundary. Kilimanjaro and the border with Tanzania lie about 50km to the south-west. It is found between latitudes of -2.37 and -2.74 decimal degrees south, and longitudes of 37.40 and 37.87 decimal degrees east.

At the time of writing, the majority of Mbirikani was communal land, owned and run by 4650 members of Ilkisongo Maasai (Ntiati 2002). There were just over 10,000 people living on the ranch, along with some 60-90,000 head of livestock (this study). In 2005-6, all land with potential for irrigation was privatised, with each member allocated 2 acres. Rough plans were in place for the remainder of the ranch to be privatised, giving each member 60 acres, and leaving some land as a conservation area.

The vegetation on Mbirikani ranged from upland grasslands to flat savannah grasslands to dense bush, and is described in detail in Chapter 2. Permanent water was scarce and restricted to a few swamps, the Kikarangot River along the southern boundary of the ranch and a water pipeline running south to north in the western quarter of the ranch. Rainfall was erratic and averaged between 350 and 500mm per year along an east:west gradient (Worden *et al.* 2003), making it one of Kenya's driest areas. The rain falls in two seasons: the short rains in November and December and the long rains from March to May (Ntiati 2002). Parts of Mbirikani provide an important wet season dispersal area for vast numbers of herbivores from Amboseli and Tsavo West National Parks and surrounding ranches (Western 1973).

1.3.2 Merueshi Group Ranch

Merueshi Group Ranch (45,201 acres) was much smaller than Mbirikani which it bordered to the north. It is found between latitudes of -2.25 and -2.39 decimal degrees south, and longitudes of 37.50 and 37.66 decimal degrees east. Merueshi was part of the South Kaputei Group Ranches (Bekure *et al.* 1991), and the Maasai living there were Kaputei Maasai.

In 1984, a decision was made by the members of Merueshi Group Ranch to privatise the land. Formal demarcation of the plots began in 1991 and was officially completed in 1997 when title deeds were issued (Merueshi chairman, *pers. comm.*). Even at the time of writing, however, not everyone held their title deed, indicating how lengthy the process of land subdivision can be. This was largely due to economic reasons; subdivision is an expensive process and communities often struggle to raise the necessary cash to pay the surveyors. Subdivision of the land occurred post privatisation as land was further fragmented and fenced. Mean plot size on Merueshi was 370.5 acres with a range between 173 and 1136 acres. There was a total of 99km of fencing in place on Merueshi at the end of 2006 (this study).

Merueshi had 117 registered members, 2000 people and some 11,000 livestock. There was no natural permanent water on Merueshi, although the water pipeline ran close to its western boundary, and there was one functioning borehole at the time of this study. One seasonal river, the Kiboko River, ran through Merueshi and during dry months people dug temporary wells in the riverbed to access water. Rainfall was similar to that on Mbirikani (350-500ml/yr), although Merueshi may get slightly more (Bekure & Grandin 1991).

Note on area units

Throughout this thesis, acres are used as the unit of area, rather than hectares (or km^2) as is more common. This is because acres are the units most familiar to the Maasai, and were used in every discussion involving land, subdivision and conservation areas. The land surveyor for the Mbirikani subdivisions worked in acres and all shambas (small agricultural plots) were demarcated in acres. The conversion factor to hectares is 1 acre = 0.4046 hectares (or 1 hectare = 2.4711 acres).

1.4 EXPLANATION AND VALUE OF THESIS

Land subdivision is an imminent challenge facing the Amboseli-Tsavo Ecosystem. It has the potential to cause the decline and even local extinction of much of the wildlife in the area. It could dramatically change the traditional pastoral way of life (for better or worse) and have a major impact on the landscape, as well as the economy of the ecosystem. Despite the importance of this issue, within the vast body of literature available on wildlife conservation and pastoralism, there seem to be very few multi-disciplinary studies focussing on land subdivision and sedentarisation and its effect on wildlife. There is a wealth of conservation/biology literature from pastoral regions, and even more anthropological studies. More recently economists have taken an interest in pastoral areas, but a combination of these disciplines is rare.

This thesis draws together all these different disciplines to produce a comprehensive, holistic overview of the changes facing the Maasai of the Amboseli-Tsavo Ecosystem, their environment and their wildlife. These changes can be both positive and negative. For many, land privatisation presents an opportunity to move away from group leadership allowing them to personally receive any revenue generated from their land. Whilst in the wetter areas of Kenya, a conversion to agriculture would be the best way to generate money from the land, in the Amboseli-Tsavo Ecosystem, and in particular Mbirikani Group Ranch, the majority of the land is too dry for agriculture to be an option. Although precluding higher earnings from agriculture for the Maasai, this is fortunate from a conservation perspective, as it is very difficult for wildlife revenues to even approach those from agriculture (Norton-Griffiths *et al.* in press). Therefore, the conservation challenge is simplified, i.e. making a mixed wildlife and livestock production system a viable and sustainable land use option. This not only requires maintaining the provision of ecosystem services required by wildlife (grazing and water resources and freedom of movement), it also requires the costs from wildlife to be better balanced by wildlife-generated revenues. Finally, it requires the input and support of the local community, which relates closely to the land use economics.

Thus the conservation of wildlife in the Amboseli-Tsavo Ecosystem during and after land subdivision is two-fold, environmental and economic. In the first instance, the land needs to be kept as open as possible, to allow wildlife to move freely between the national parks and wet season dispersal areas on the group ranches. Secondly, the local community members need to benefit significantly from wildlife in order to make it in their interest to protect it or even continue to tolerate it. This thesis aims to tackle both these issues, focussing first on the ecological effects of land subdivision, and second on the underlying socio-economic context, with the ultimate aim of suggesting a conservation plan for the future of Mbirikani Group Ranch.

1.5 THESIS STRUCTURE

Chapter 2 demonstrates some of the ecological effects of land subdivision and sedentarisation, through the comparison of Mbirikani and Merueshi Group Ranches. Data are presented in the context of a long term dataset showing the changes since pre-subdivision. The hypothesis tested is that '*Land subdivision and pastoralist sedentarisation negatively affects environmental processes and leads to a decrease in wildlife, and to a lesser extent livestock*'.

Chapter 3 continues looking at the environmental effects of subdivision by investigating to what extent subdivision affects the distribution of wildlife. This chapter tests the hypothesis that '*Land subdivision and sedentarisation of Maasai pastoralists compromises the ability of wild grazers to distribute themselves optimally within the landscape*'.

Chapter 4 takes an economic angle. It tests the hypothesis that, '*For a Maasai pastoralist in the Amboseli-Tsavo Ecosystem, the cost of living with wildlife greatly exceeds income from current wildlife revenues*'. It quantifies the costs incurred by a Maasai household due to the presence of wildlife on their land, and determines the extent to which these are offset by wildlife revenues. It explains the importance of generating sufficient wildlife-related benefits before the community can be expected to act in a pro-conservation manner during the land subdivision process.

Chapter 5 follows on from this with an assessment of community attitudes towards wildlife and its conservation. It tests the hypothesis that '*The presence of wildlife revenues positively influences pastoralist's attitudes to wildlife, but are currently insufficient to create behavioural change*'. It discusses the extent to which behavioural change is brought about by the receipt of income from wildlife and suggests ways in which revenue could be better managed and distributed.

Chapter 6 summarises the main findings of the thesis within a global context and discusses their relevance to other community based conservation efforts around the world. It also summarises the current literature regarding conservation planning.

Chapter 7 concludes the thesis with a conservation plan for Mbirikani Group Ranch based on findings from the study and the conservation planning literature discussed in Chapter 6.

CHAPTER 2

THE ENVIRONMENTAL EFFECTS OF LAND SUBDIVISION AND INCREASING SEDENTARISATION OF SEMI-NOMADIC PASTORALISTS

Hypothesis: Land subdivision and pastoralist sedentarisation negatively affects environmental processes and leads to a decrease in wildlife, and to a lesser extent livestock.

ABSTRACT

Pastoralists around the world are becoming more sedentary and communal rangelands becoming privately allocated. Land allocation and titling is now inevitable in the communal Maasai areas of southern Kenya. The next step, physical subdivision of the land through fencing and other barriers, has occurred in many Maasai ranches and will potentially follow land allocation in the remaining areas. In this chapter, quantitative data are presented to illustrate the decline in wildlife populations and deterioration of range condition resulting from land subdivision and sedentarisation of semi-nomadic Maasai. Two Maasai ranches were compared, one communal and one subdivided. There were significantly lower wildlife densities on the subdivided ranch, although long-term data showed that there was no significant difference prior to land subdivision. The subdivided ranch also had significantly lower grass ground cover and grass biomass than the communal ranch which historically used to have less grass. The results suggest that land subdivision should be avoided to reduce loss of the vegetation and wildlife resource.

2.1 INTRODUCTION

2.1.1 Background

Land allocation refers to the legal division of communal lands into private land parcels, with title deeds issued to individuals. This is now inevitable in the remainder of Kenya's Maasailand but can be done in many different ways. The physical division of the land, through boundary markers, fences and enforced property rights, henceforth known as land subdivision, frequently follows official land allocation (BurnSilver & Mwangi 2007). The historical context for land allocation and land subdivision, and some of their broader implications, are discussed in Chapter 1. This chapter will focus on the ecological significance of land subdivision and one of its major consequences, sedentarisation of semi-nomadic pastoralists. Specifically, the following hypothesis will be investigated.

"Land subdivision and pastoralist sedentarisation negatively affects environmental processes and leads to a decrease in wildlife and to a lesser extent livestock."

Sedentarisation refers to the settlement of previously nomadic or semi-nomadic people into permanent homesteads with a corresponding decrease in the mobility of people and their livestock (Salzman 1980). There is a wealth of reasons why pastoralists may choose to settle including economic, political, demographic and environmental changes, especially drought (Njoka 1979; Roth & Fratkin 2005). In Kenya for example, increasing population pressure, continued loss of rangelands to non-pastoral sectors and development interventions in pastoral economies have contributed to a rapid decrease in the mobility of pastoral herds and households throughout the country (Fratkin 1992; Schwartz *et al.* 1995; Roth 1996). Land subdivision is often considered one of the main, but not the only, driving forces for sedentarisation (Graham 1988).

Social repercussions of subdivision and sedentarisation can be devastating, and have been comprehensively reviewed (Galaty 1992; Kimani & Pickard 1998; Thornton *et al.* 2006). The change to a semi-sedentary herding system can result in the breakdown of social structures which serve as a social security system within these pastoral communities (Schwartz 2005). In addition, land subdivision can result in previously accessible pastures becoming unavailable (Homewood 1995) and livestock losses to drought increasing substantially (Scoones 1992; Boone 2005). Food insecurity may result (Thornton *et al.* 2006), and the sale of land and resulting landlessness can also have major social repercussions (see Galaty 1992).

However, the focus of this chapter is on the *ecological* effects of subdivision and sedentarisation. In a recent survey, land subdivision was consistently declared the greatest threat to the long term sustainability of livestock, human and conservation interests by Maasai group ranch members, Kenya Wildlife Service representatives and research scientists (Boone *et al.* 2005). Stanley (2000) writes "Rangeland destruction due to overstocking is further aggravated by subdivision into plots too small to survive even as subsistence holdings. The wildlife in these areas disappears and the lands become so overgrazed that nothing can be productive". One aim of this chapter is to investigate the truth of this statement for a ranch in Kenya's Kajiado District and to discuss the extent to which it is an inevitable result of future subdivisions.

Many studies report negative environmental consequences of land subdivision and sedentarisation (e.g. Stanley 2000; Ntiati 2002; Seno & Shaw 2002; Worden *et al.* 2003; Schwartz 2005). For example, in the Laikipia District of Kenya, wildlife numbers and diversity are significantly lower on smaller holdings as compared with larger ranches (Norton-Griffiths 1998), and fencing of the land may cause migrations to be rerouted and lead to exhaustive grazing pressure in a confined space (Schwartz 2005). Boone *et al.* (2005) modelled the changes in livestock herds due to land subdivision and found that, for ranches of low productivity, there was a steady decline in the capacity of the land to support livestock under subdivision. This would inevitably apply to wild macro herbivores as well (Boone *et al.* 2005).

This chapter investigates the environmental effects of land subdivision on Merueshi Group Ranch, a small Maasai ranch in southern Kenya which subdivided in the 1980s. Evidence from the early-mid 1970s (presented in this chapter) suggests that prior to subdivision Merueshi Group Ranch was ecologically similar to the neighbouring Mbirikani Group Ranch, which is used here as a comparison to illustrate the changes which have occurred on Merueshi since subdivision. Whilst this particular research project was too short to measure the actual changes since the 1970s, long term data collected by Dr David Western provides the background to what is presented here (D. Western, unpublished data).

2.1.2 Case study: the subdivision of Merueshi Group Ranch

By the early 1980s, people on Merueshi Group Ranch were already beginning to show signs of reducing their nomadic ways. During this period, more households were sedentary in Kaputei than on Mbirikani (Grandin, de Leeuw & ole Pasha 1991), and by 1981 more than 90% of Kaputei household heads were living in their permanent boma, as compared with only 46% of household heads on Mbirikani (Grandin *et al.* 1991). The official privatisation of Merueshi began in 1984 and was completed in 1997. Plot sizes on Merueshi at the time of study ranged from 0.7 to 4.6km^2 (mean 1.5km^2).

2.1.3 Case study: Land tenure on Mbirikani Group Ranch

At the time of writing the majority of Mbirikani, with the exception of the irrigable land, remained communal, although there were plans to privatise the remainder of the land. These entailed leaving a section of the ranch at the base of the Chyulu Hills as a communal grazing and wildlife conservation area, and subdividing the remainder of the dry rangelands into 60 acre plots

2.1.4 Research Objectives

In order to investigate the hypothesis that subdivision negatively affects environmental processes using a comparison of a subdivided and communal ranch, it is necessary to first demonstrate that the two areas were ecologically similar and part of the same greater ecosystem prior to subdivision. Then current differences between the ranches can be investigated, and the extent to which they can be considered a consequence of land subdivision discussed. In the light of this, this chapter has the following specific objectives:

- To provide evidence that wildlife production was similar on Mbirikani and Merueshi Group Ranches prior to the land subdivision of Merueshi.
- To describe the current landscape on both ranches, including habitat classification and the distribution of major features such as rivers and roads.
- To compare characteristics of the current grass resource on the two ranches.
- To compare the current densities of wildlife and livestock on Mbirikani and Merueshi.
- To illustrate the mobility of the pastoralists by describing the distribution and abundance of both permanent and temporary bomas.

2.2 METHODS

2.2.1 Use of long term data

The Amboseli Research and Conservation Project has collected ecological data in the areas covered by this study since 1974. These data (D. Western, unpublished data) made possible a direct comparison of long term changes of wildlife and livestock production between Mbirikani Group Ranch and the Kaputei Ranches, of which Merueshi is a part. The data for this comparative study comes from an aerial sampling program established by D. Western in 1973, using a grid system designed to count and monitor large mammal populations in the Greater Amboseli Ecosystem. As in this study, all macro herbivores above Thomson's gazelle in size were counted. Animal production per unit area was used as the unit of measurement for comparing between ranches. Animal production was calculated for each species using the equation $P=N\ 13.8\ M_s^{0.67}$, where N is the population size or density and M_s is the mean kcal equivalent of adult mass (Western 1983). Unit weights were based on values given in Western (1973), and calories are used rather than joules to allow the Western (1983) method to be followed directly. For information, since joules may be more familiar, the conversion factor is 1 calorie = 4.1840 joules (or 1 joule = 0.2390 calories).

The mass scaling exponent of 0.67 used to calculate production is necessary to account for the different food requirements of different sized species, in turn due to different rates of energetic loss resulting from different volume to surface area ratios (Demment & Van Soest 1985). There is an ongoing debate in the literature about exactly what this scaling exponent should be (White & Seymour 2005; Clauss *et al.* 2007), but to calculate production, I use the method developed by Western (1983), with the exponent of 0.67, as this has been used as the standardized method for calculation of production in the Amboseli Ecosystem for several decades (Western 1989,1991; Western, Russell & Mutu 2006) and is based on measured turnover rates. Results from my study would thus be directly comparable with these studies. However, later in this thesis (Chapter 4) I follow a different methodology which uses a scaling factor of 0.75. The reasons for the difference between these scaling factors are discussed in Chapter 4, section 4.2.5.1.

In this chapter, I have drawn on Western's long term data as a background for the situation observed in 2005 on Mbirikani and Merueshi Group Ranches. Since Merueshi was so small in comparison with Mbirikani, for the aerial sampling data, the whole of the Kaputei region was used for comparative purposes. However, for Merueshi on its own to be considered representative of Kaputei it was necessary to compare the wildlife and livestock production in the two areas to ensure there were no significant differences. Due to the size difference between Merueshi and the remainder of Kaputei (7 versus 44 grids in the aerial sampling design), a Monte-Carlo re-sampling technique was used. Essentially, the mean and median of the 7 Merueshi grids were compared with those of 10,000 randomly selected sets of 7 grids from Kaputei, using Mann-Whitney U-Tests, and the percentage of times the results were significantly different recorded. If they were found to be significantly different more than 5% of the time, the areas were considered significantly different overall.

2.2.2 Habitat classification and vegetation surveys

A 2002 georeferenced LANDSAT image was used to identify the 11 major habitat types present on the ranches (Oindo, Skidmore & De Salvo 2003). Ground truthing was carried out at over 200 randomly chosen locations. Vegetation density (no of trees and shrubs per km^2) was recorded by vegetation transects. In each of the 11 habitats, six transects were done at randomly chosen locations (Eccard, Walther & Milton 2000). Each transect was 60m in length and all plant species (excluding forbes and grasses) within one meter to the right of the tape measure were recorded. Any unknown species were coded and a specimen taken for later identification. The books used to identify specimens included Kenya Trees, Shrubs and Lianas (Beentje 1994) and Upland Kenya Wild Flowers (Agnew & Agnew 1994), and expert botanical help was provided by a retired botanist from the Kenya Agricultural Research Institute.

In addition, in each habitat, a minimum of 20 grass quadrats were done at randomly chosen locations. The quadrat was 1m x 1m and all grass species present within it were identified and their percentage cover recorded (McLaughlin & Bowers 2006). The following books were used to identify samples: An illustrated manual of Kenya grasses (Ibrahim & Kabuye 1987) and A revised list of Kenya grasses (Bogdan 1976). Appendix 2A gives detailed results.

2.2.3 Livestock and wildlife censuses

Animal population counts were done monthly using strip transects (Burnham, Anderson & Laake 1980; Sutherland 1996). This technique has frequently been used to estimate mammal densities (Vidal *et al.* 1997; Caro 1999a; Dique *et al.* 2004). Every month, for Mbirikani Group Ranch, a minimum of 22 strip transects of 4km in length were laid out in a stratified random sampling design according to habitat (Krebs 1999) and further stratified by wildlife abundance. The transects were different each month, and start and end points and orientation while driving were determined with a Global Positioning System (GPS III+, Garmin). Speed never exceeded 15km hr^{-1} and the driver and single observer were the same for all transects. Monthly transects took 3 full days and were usually completed within the first week of each month. Transects were carried out throughout the day, but the order in which they were done was randomised each month to avoid bias from differences in animal visibility at different temperatures. In total 1132km (283 transects) were driven.

The maximum distance at which one can guarantee to see a Thomson's gazelle (the smallest animal to be counted) was chosen as the strip width on either side of the car (see Caro 1999a). This could be different for each transect and was modified during the transect if vegetation density and hence visibility changed (Western 1973). At every sighting, the animal or groups' exact GPS position was recorded, as well as its distance and angle from the vehicle. This was done using a digital range finder (Yardage Pro 500, Bushnell Sports Optics Worldwide) and an angle board (Buckland *et al.* 2001 pp. 263). Group size was also recorded.

The area of each transect driven was calculated by multiplying width by length for all the different sections of one transect and then summing them (Burnham *et al.* 1980). Species density was calculated by summing the total number of individuals of that species seen on the transect and then dividing by the area visible (Mduma 1995; Caro 1999a). Caro (1999b) found that this method gives densities that are strongly correlated with densities obtained through other ground-based methods.

Point transects were used where line transects were not possible. This included the boulder field habitat on Mbirikani (due to inaccessibility of the terrain by vehicle), and the whole of Merueshi Group Ranch (due to fences and private land ownership). Fifty point transects were done each month on Merueshi Group Ranch and 25 within the boulder field habitat of Mbirikani. The point from which the count was done was chosen randomly and accessed by bicycle or foot. The radius of the circular area to be counted was chosen according to visibility, and measured with the range finder. Once in position, the observer remained still for three minutes before beginning the count to allow animals time to settle down and resume original behaviour. Data were recorded as for strip transects.

Point transects use the same concept as belt transects, where all the animals within a certain fixed area are counted (Sutherland 1996), and have been shown to give very similar density estimates to strip transects (Ruette, Stahl & Albaret 2003; Guidetti *et al.* 2005). Nonetheless, a comparison test was done to determine whether density estimates from point or line transects differed significantly or not in this study. In three different months (March, October and November), a series of independent belt and point transects were conducted in two different habitats, open grassland and thinly bushed grassland. As the data were not normally distributed, they had to be analysed using a series of non-parametric (Mann-Whitney) tests for each species (N=12) in each month-habitat combination (N=6) i.e. N=72 tests in total. Although this involved a multiple testing procedure, the significance level was retained as $\alpha = 0.05$ to minimize Type II errors. The results are given in Appendix 2B.

For certain analyses, wildlife were grouped into feeding guilds. Wild grazers included zebra, wildebeest, Thomson's gazelles, oryx and Coke's hartebeest; wild browsers included gerenuk, giraffe and lesser kudu, and wild mixed feeders included eland, Grant's gazelle and impala. For analysis of all transect data, non-parametric statistics in SPSS (version 12.0) were used because the large number of zeroes in the data set precluded use of parametric statistics (Caro 1999a). For investigation of grazing pressure, wildlife production was used (Western 1983) rather than density, as it represents the forage offtake by herbivores, rather than simply how many herbivores there were per unit area (see section 2.2.1 for calculation).

2.2.4 Grass sampling

Grass characteristics were measured every 500m along each 4km strip transect, and twice in the area surveyed at each point transect. This was done using the pin intercept (point frame) method (Sutherland 1996; Mwangi & Western 1998), used as the standard plotless method for measuring grass characteristics in countless studies (Wilen & Holt 1996; Mwangi & Western 1998; Shaver *et al.* 2001). In this method, a wooden A-frame supports ten metal pins of one metre in length, angled at 33^0 to the vertical. This pin-frame is placed at a randomly selected site at each sampling point. The number of grass blades touching each pin is recorded and then the total divided by ten to get a score of 'mean blades per pin'. This measure can be directly correlated with biomass, once calibrated by measurement of clipped plots (Mwangi & Western 1998). In this study 104 clipped plots and pin-frame pairs were used to calibrate the pin-frame. Calibration involved cutting, air-drying and weighing all grass within a $50cm^2$ quadrat at the site of the pin frame measurement. The resulting weights (in grams) were multiplied by four to get a measure of biomass in g/m^2. The resulting biomass scores were plotted on a graph against the original recordings of mean number of blades per pin, and a regression line fitted with the intercept forced through zero, since it is biologically impossible to have negative biomass. The equation of the regression line was used to transform all measurements of mean blades per pin into biomass. Above-ground biomass was used in order to give the best indication of the biomass available to grazing herbivores (McNaughton 1985).

Every grass blade that touched a pin was also recorded as green or not-green and grazed or not-grazed. The numbers of grazed or green blades were divided by the total number of blades counted and multiplied by 100, to give an estimation of the percentage greenness and grazing (Western 1973). The step-point method was used to measure ground cover (Strauss & Neal 1983; Sutherland 1996). Fifty steps were taken in a perpendicular direction from the car and each time the toe of either foot touched grass, it was marked down. The total number of positive scores was multiplied by two to get an estimate of percentage ground cover. This was done twice at each sampling point and the values averaged, in an effort to avoid bias.

2.2.5 Classification of months into seasons

In order to make comparisons between ranches, both animal and grass data were averaged by season. Months were divided into seasons on the basis of grass greenness and grass biomass (Mwangi & Western 1998; Mose 2005). Percentage greenness and percentage deviation from the overall biomass mean were used to classify the months initially into wet, dry or drought seasons. Following the methodology of the African Conservation Centre for their Amboseli project, any months with grass having 25% green or above were classified as wet. Below 25% green, it was the biomass of grass available that determined season. Months with a 0 to -50% biomass deviation from the overall biomass mean were classified as dry season months, while a -50 to -100% biomass deviation was classified as a drought month (see Mose 2005). Any months that did not fit into any category based on this system were classified by visually assessing the data and choosing the most appropriate season. Since there was only one month (October) falling into the drought season classification, this was re-classified as dry season, meaning data were ultimately grouped into only two seasons, wet and dry. Ultimately, January, February, April, May, June, November and December were classified as wet months, and March, July, August, September and October classified as dry. Details of these results are presented in Appendix 2C.

2.2.6 Boma and human population survey

A survey of all the permanent bomas on both ranches was carried out in 2005. A permanent boma was defined as one which had been standing at its present site for over three months and in which there were always some family members and a few head of livestock, even when the bulk of the herds were moved out into temporary bomas. The Maasai term for these permanent bomas is *emparnat*, and this term was used during the surveys to avoid confusion. The position of each *emparnat* was recorded using a GPS unit. In addition to recording the permanent bomas, once per season (wet and dry) in 2004, 2005 and 2006, the GPS positions of all temporary bomas were recorded. A temporary boma was defined as one which a household was using in a transitory fashion, for less than three months. All surveys were carried out by one of two trained Maasai enumerators and respondents were any adult (>16 years) present at the boma at the time of the visit. The survey was short and simple and asked about the number of men, women and children who would sleep in the boma that night, in order to estimate the human population densities on the ranches. Once all the boma positions had been recorded, an index of dispersion was used to investigate the spread of the bomas throughout the ranch (Fowler, Cohen & Jarvis 1998).

2.3 RESULTS

The majority of this chapter presents results from field research carried out in 2005. First however, results of a long term sampling project are presented as a background to the snapshot situation illustrated by the 2005 results

2.3.1 Long term comparison of Kaputei and Mbirikani

Results (from Western's long term aerial sampling program) were analysed by decade, with counts from 1974 to 1979 (n=17) representing the 1970s, counts from 1980 to 1989 (n=6) representing the 1980s and all counts post 1990 (n=6) representing the 1990s-2000s. Figures 2.1 and 2.2 summarise the changes by decade of wildlife and livestock production respectively on both Mbirikani and Kaputei. Wildlife production was very similar in the 1970s and 1980s on both ranches (T=0.006, P=0.998 and T=0.061, P=0.953 respectively), but by the 1990s-2000s wildlife production on Kaputei was significantly lower than on Merueshi (T=2.709, P=0.042). For livestock however (see Figure 2.2), there was a significant difference in production in the 1970s (T=-4.007, P=0.001) but this was no longer significant by the 1980s or post 1990s. This result should be interpreted cautiously however, since the large variances in the latter two decades (due to the smaller sample size) may be the reason for the apparent lack of significance.

Figure 2.1 Decadal trends in wildlife production on Mbirikani (solid line) and Kaputei (dashed line) between May 1974 and March 2006. Means ± standard error bars (D. Western, unpublished data).

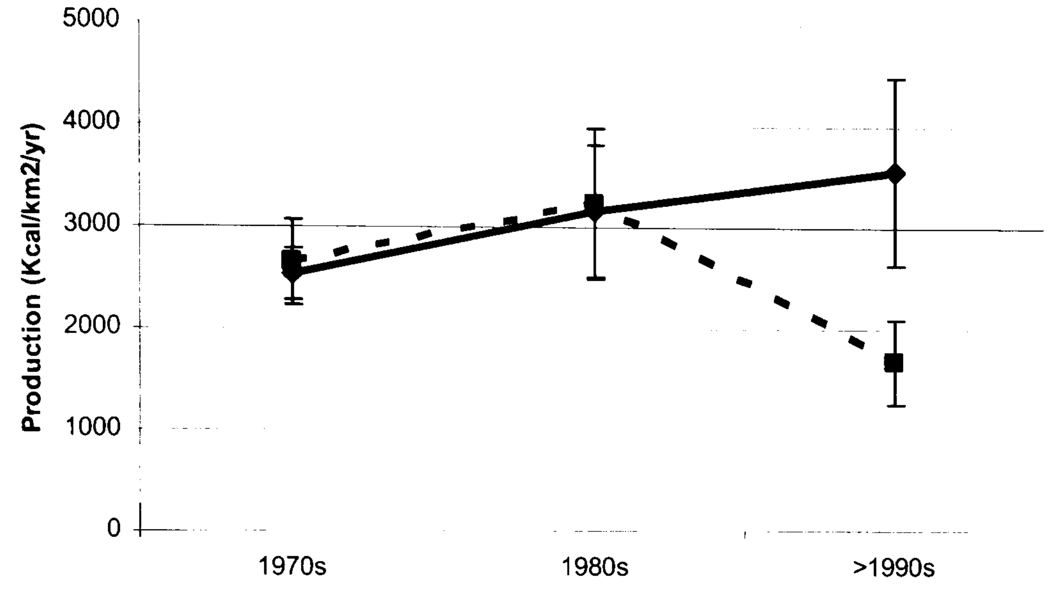
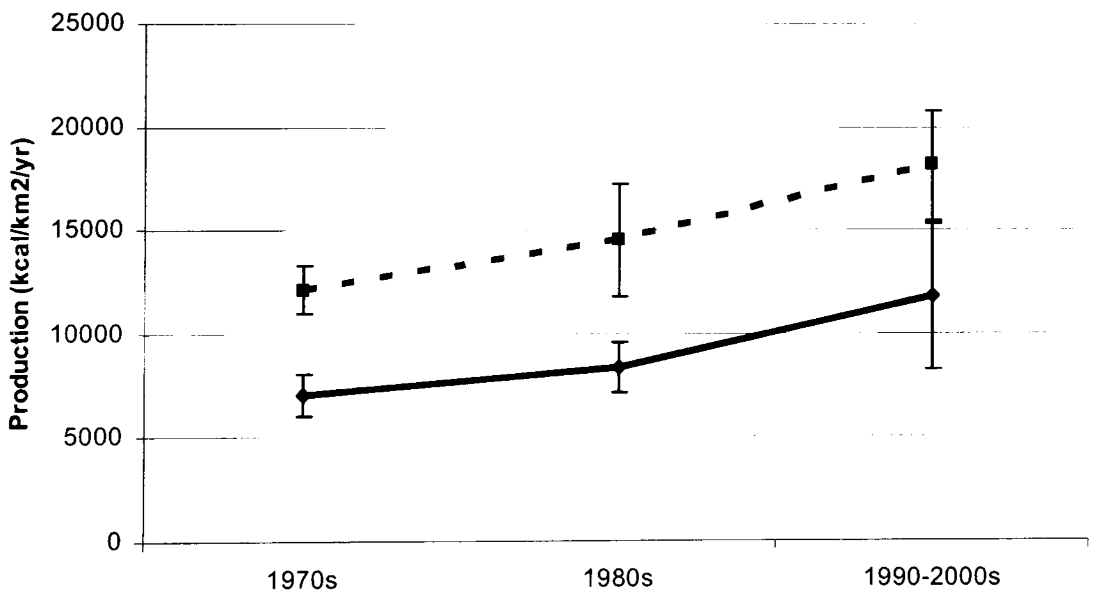


Figure 2.2 Decadal trends in livestock production on Mbirikani (solid line) and Kaputei (dashed line) between May 1974 and March 2006. Means ± standard error bars (D. Western, unpublished data).



2.3.1.1 Comparison of Merueshi and Kaputei

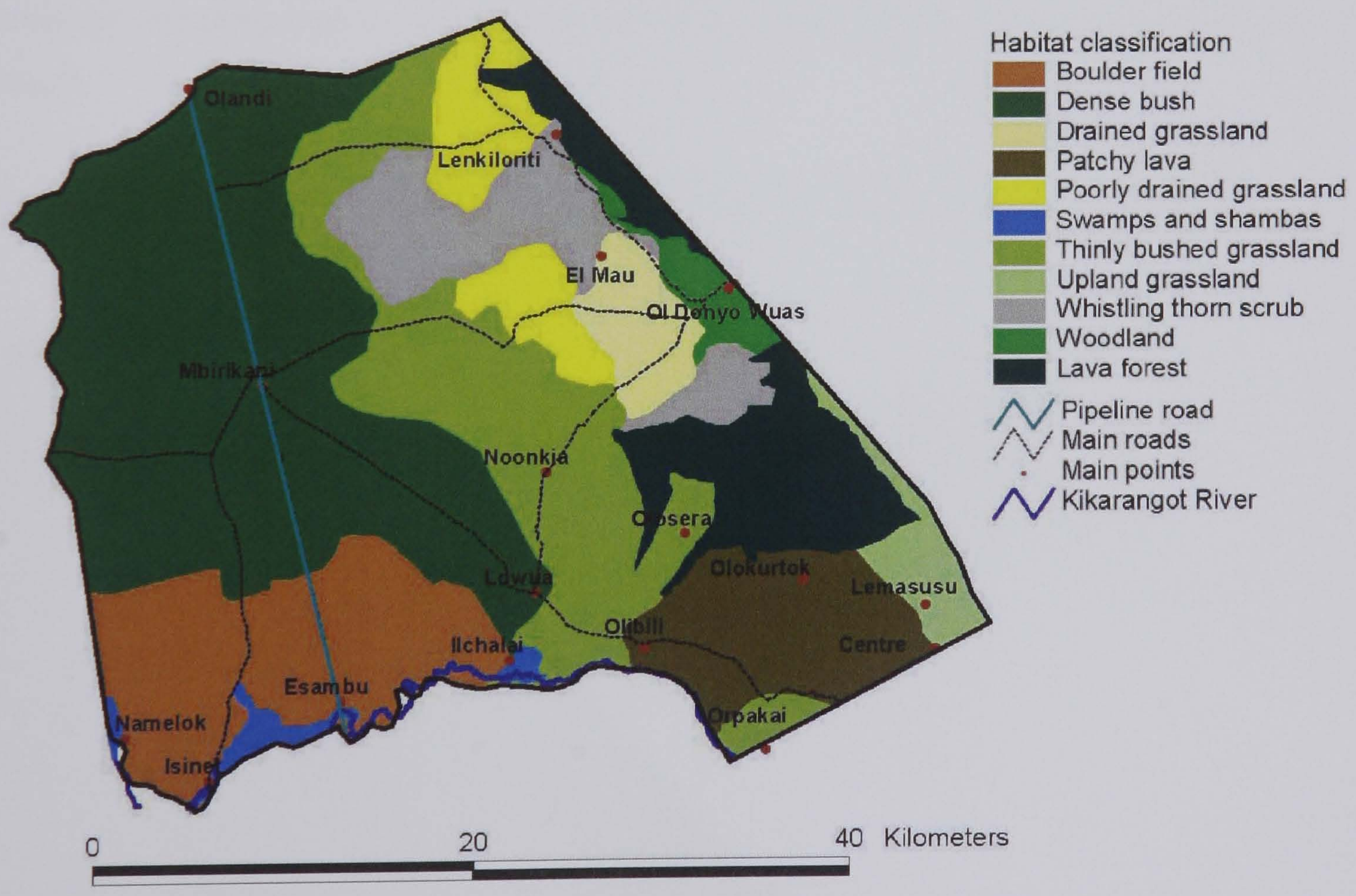
Results of the re-sampling technique comparing Merueshi and Kaputei showed that for wildlife production, there was no significant difference between the two areas; the proportion of times a significant difference was found was less than 5% (means, P=0.027;

medians, P=0.018). This was applicable even when wildlife was broken down into feeding guilds (grazers, browsers and mixed-feeders); in all cases P<0.05 for both means and medians. For livestock however, results did show a significant difference between Merueshi and Kaputei; the percentage of the 10,000 random samples that gave a different result was more than 5% (means, P=0.151; medians, P=0.170). From this I conclude that for wildlife, Merueshi is a fair representation of Kaputei, but for livestock there are some significant differences.

2.3.2 Description of ranches

Figures 2.3 and 2.4 illustrate the different habitat types present on Mbirikani and Merueshi Group Ranches in 2005. Whilst Mbirikani had many more habitat classes than Merueshi, the two dominant categories were the same, classified in this study as 'dense bush' and 'thinly bushed grassland'. A full description of each habitat type is given in Appendix 2A.

Figure 2.3 A map of Mbirikani Group Ranch showing the dominant habitat types, main roads and rivers.

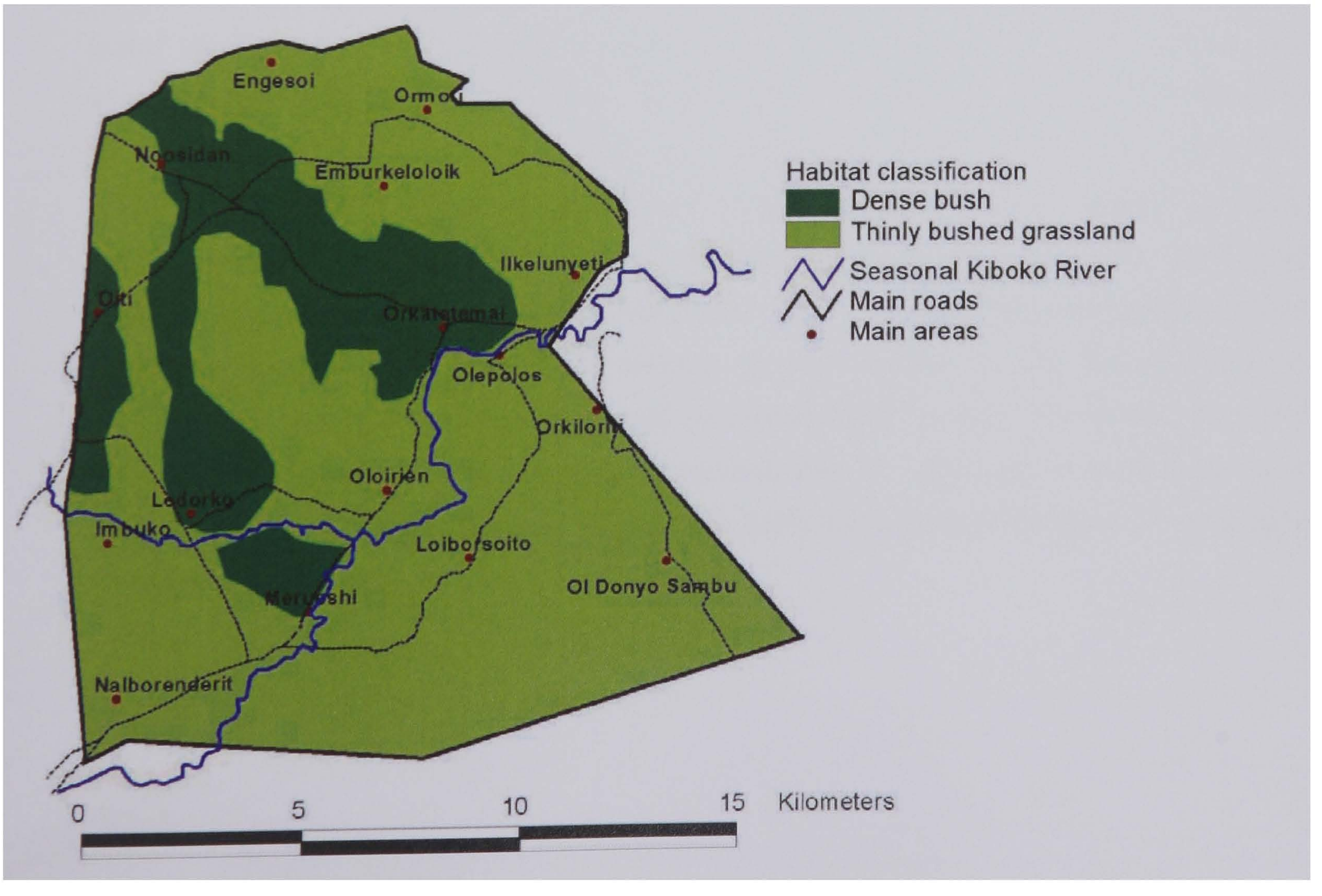


On Mbirikani, the density of the vegetation in each habitat decreased from west to east, with the exception of the eastern lava forests which comprise very thick vegetation. The dense bush in the west of the ranch comprised stands of *Commiphora rostrata* and mixed

Acacia-Commiphora woodland. The grassland plains in the east were virtually devoid of any shrubs or trees and were a favoured grazing area for both wild and domestic herbivores, with dominant grass species being *Pennisetum mezianum* and *Chrysopogon aucheri* on the poorly drained soils, and *Sporobolus pellucidus* and *Digitaria milanjiana* in the well drained grasslands. To the east of Mbirikani are the Chyulu Hills National Park, where the habitat was upland grassland, dominated by *Hyperhannia hirta* and *Themeda triandra*.

The Kikarangot River on the southern boundary of the ranch is a perennial river, although it frequently dried up before reaching its end at Orpakai dam. The river was heavily utilised upstream of Orpakai, mainly for irrigation of small-scale agricultural plots (shambas). This river and the swampy areas around the shambas were the only permanent natural water sources on the ranch. The pipeline road however, which was constructed to take Kilimanjaro melt water to the towns near the city of Nairobi (Ntiati 2002), had water access points at regular intervals which provided people with water for household purposes and for watering their livestock. The pipeline was frequently leaking however, meaning that in reality there was often water widely available along much of its length.

Figure 2.4 A map of Merueshi Group Ranch showing the dominant habitat types, main roads and rivers.



Merueshi Group Ranch was dominated by thinly bushed grassland, with patches of dense bush (mainly thickets of *Commiphora rostrata* and *Cordia gharaf*). It had no lava forests, open plains or shambas, although the south east is bordered by the upland grasslands of the Chyulu Hills, as for Mbirikani. The thinly bushed grassland did contain some whistling thorn (*Acacia drepanolobium*), although this was more dispersed and not clumped into stands large enough to constitute a separate habitat, as it did on Mbirikani. *Acacia senegal* and *Acacia mellifera* were the other dominant woody species. *Pennisetum mezianum* and *Cenchrus ciliaris* were the dominant grasses in the thinly bushed grasslands. The Kiboko River which runs through Merueshi was seasonal and flowed only during good rains. The remainder of the time it was dry, although people dug wells in the riverbed to access water. There was no permanent natural water on Merueshi, although there were a few boreholes and the water pipeline was at maximum 6km from its western boundary.

2.3.3 Distribution of bomas

Figure 2.5 shows the distribution of both permanent and temporary bomas on Mbirikani Group Ranch. The permanent bomas (*emparnat*) showed a highly clumped distribution (χ^2_{86}=1290.9, $P<0.001$), as they were clustered around the permanent water sources; the water pipeline and the Kikarangot River in the south (see Figure 2.3). There were approximately 450 permanent bomas on Mbirikani, comprising around 930 households (2.1 households per boma). Dry season temporary bomas were found mostly within 10km of the pipeline road (or other source of permanent water), whilst wet season temporary bomas tended to be found mostly in the east of the ranch at the base of the Chyulu Hills. During one dry season boma survey (September 2004), 96 temporary bomas were located (indicated in Figure 2.5). Other dry season surveys located 31, 53 and 47 temporary bomas. The wet season boma survey of December 2004 located 125 temporary bomas (indicated in Figure 2.5). Another survey located only 71. Although the numbers varied, the distribution of these temporary bomas was fairly consistent and Figure 2.5 represents the standard pattern.

Figure 2.5 The distribution of bomas on Mbirikani Group Ranch

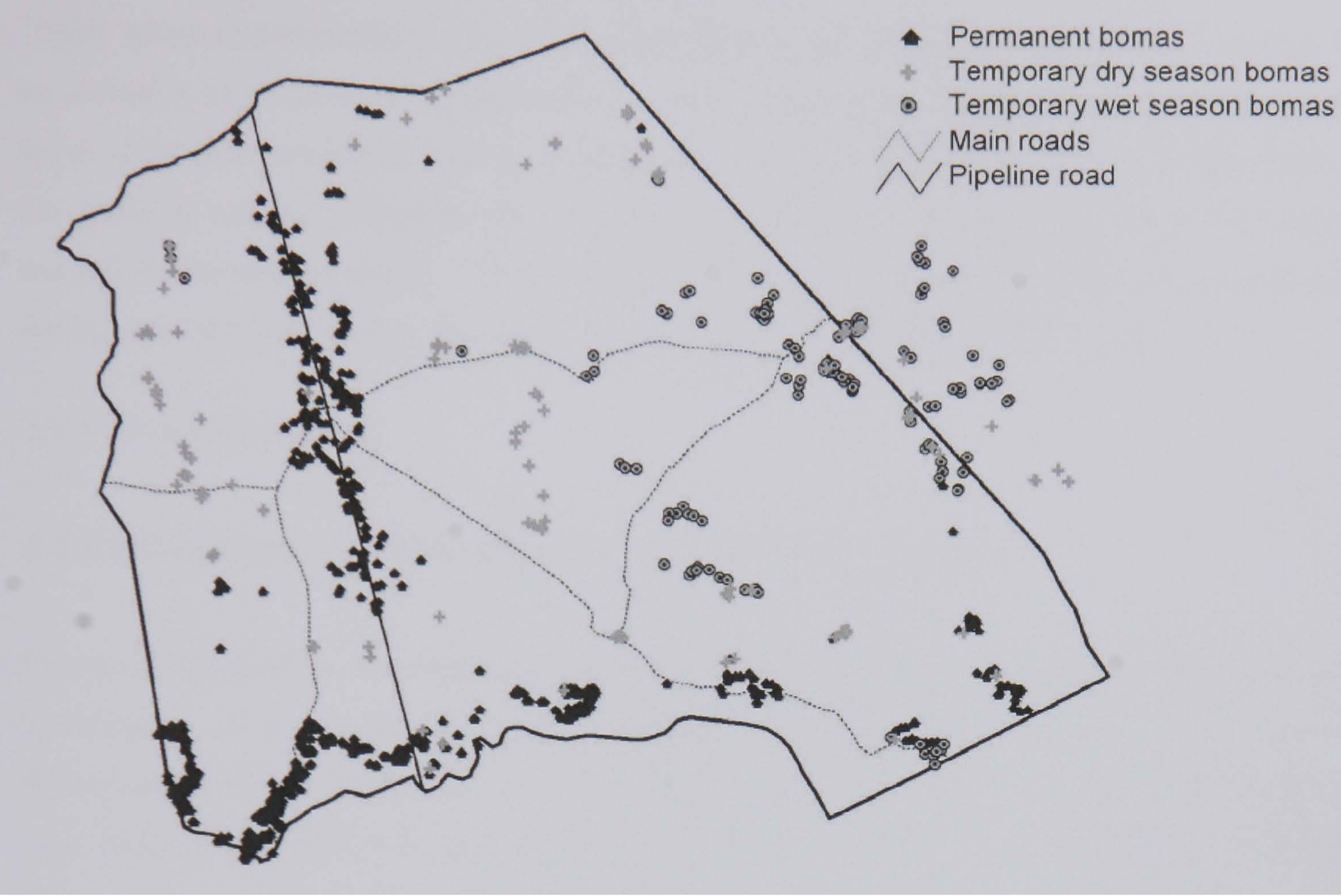
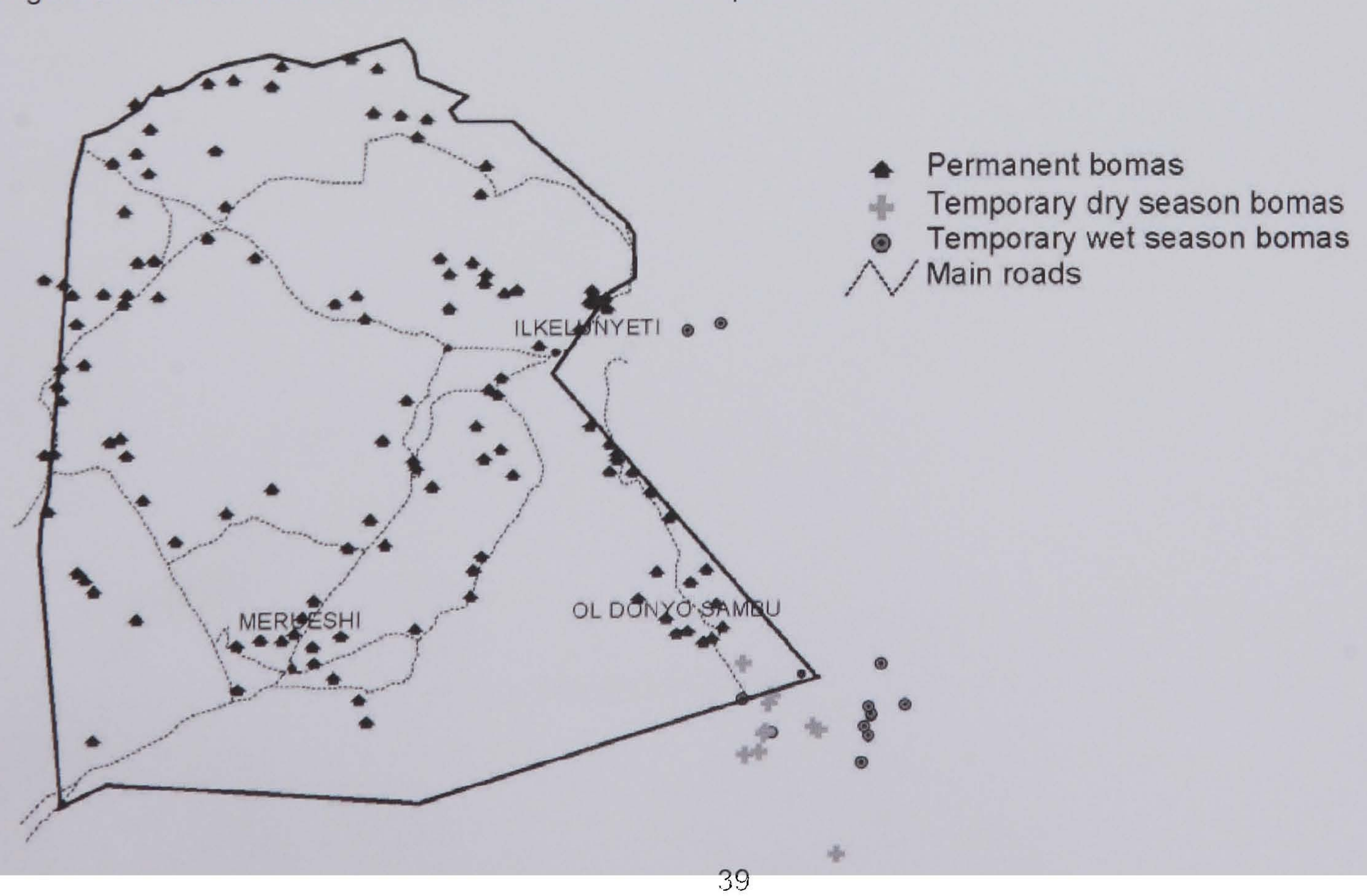


Figure 2.6 illustrates the distribution of permanent and temporary bomas on Merueshi Group Ranch. The permanent bomas on Merueshi, like Mbirikani, showed a clumped distribution (χ^2_{52}=104.3, P<0.01). However, if the concentration of bomas around the three main towns (Ilkelunyeti, Merueshi and Ol Donyo Sambu) were excluded, the remaining bomas (79%) showed a random distribution (χ^2_{49}=63.1, P>0.05).

Figure 2.6 The distribution of bomas on Merueshi Group Ranch



There were approximately 120 permanent bomas on Merueshi, comprising around 160 households (1.3 households per boma). From a total of six temporary boma surveys, only twice were any temporary bomas of Merueshi members found on or close to the ranch. A dry season survey (October 2005) located 12 temporary bomas, as did a wet season survey in November 2005. These were all clustered in the south east corner and were mostly on Mbirikani Group Ranch or Chyulu Hills National Park (Figure 2.6).

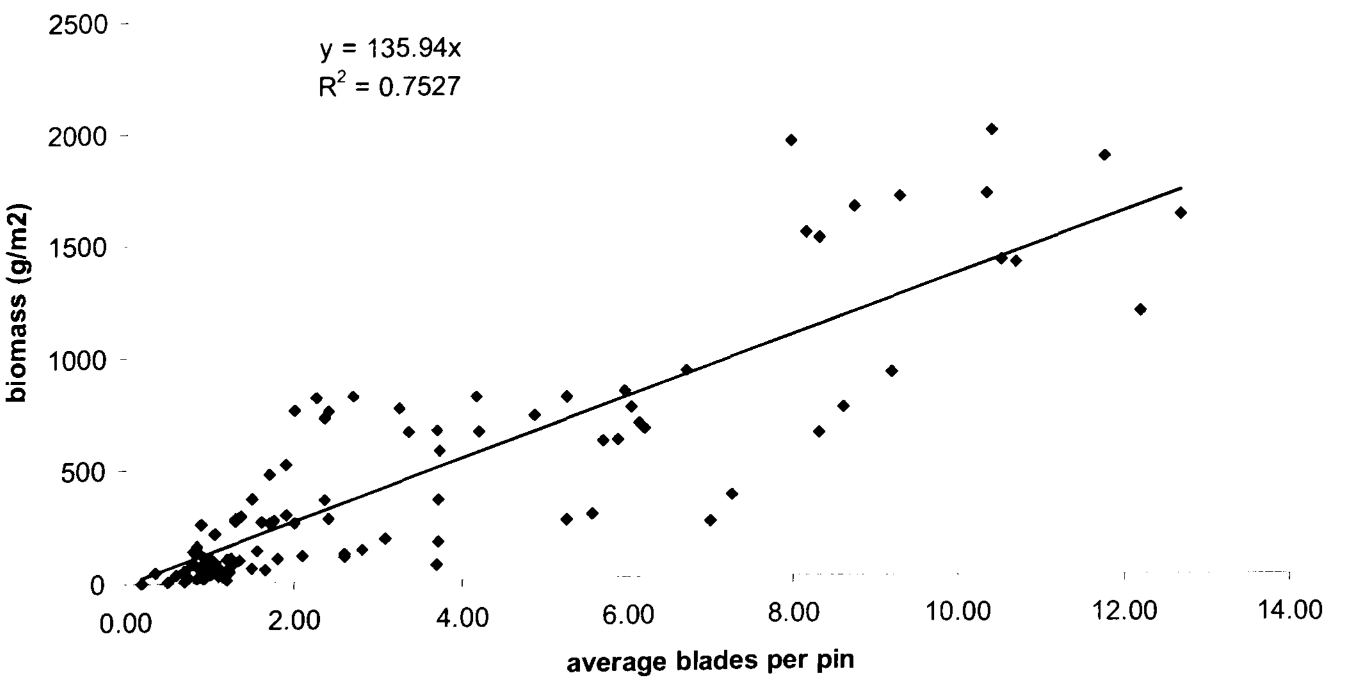
2.3.4 Grass sampling

2.3.4.1 Calculation of grass biomass from pin frame sampling

Figure 2.7 illustrates the regression equation for the calibration of the pin-frame sampling technique. The measured biomass values were plotted against their corresponding measure of 'mean blades per pin' from the pin-frame (N=104). A linear regression line was fitted to the data and the intercept set as zero. The resulting equation (y=135.94x) was used to transform all measured values of mean blades per pin into biomass, where x=mean blades per pin.

A linear regression was performed on the untransformed data, and this produced a highly significant result ($F_{1,103}$=635.79, P<0.001), showing a strong correlation between 'blades per pin' and biomass.

Figure 2.7 Comparison of mean blades per pin and grass biomass, N=104.



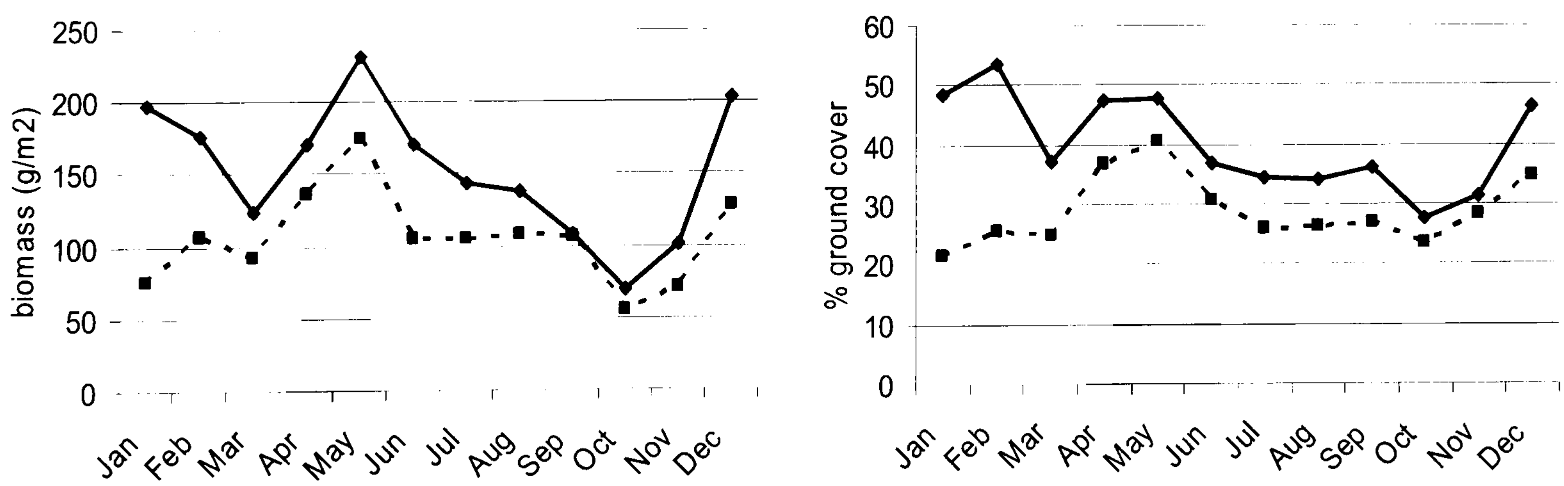
2.3.4.2 Differences in grass characteristics between the two ranches

The hypothesis that land subdivision negatively affects environmental processes can be tested by investigation of the following sub-hypothesis:

"Sedentarisation, relative to nomadic, seasonal use of the rangelands, will result in less plant cover, lower biomass and less seasonal flux in both of these."

Figure 2.8 shows the monthly changes in grass biomass and ground cover on Mbirikani and Merueshi. Mean monthly biomass on Mbirikani was consistently and significantly greater than Merueshi biomass (paired samples t-test: t_{11}=5.021, P<0.001) with the greatest difference being after the short rains and before the long rains (December to February). Monthly percentage ground cover by grass was also significantly higher on Mbirikani than Merueshi (paired samples t-test: t_{11}=4.855, P=0.001), both of which support the sub-hypothesis above. Grass biomass on both ranches showed temporal fluctuations which correlated significantly in a positive direction with the previous month's rainfall (Pearson's correlation: R=0.777, P=0.003 and R=0.657, P=0.020 for Mbirikani and Merueshi respectively). However, the variance of grass biomass on Mbirikani (2258.44) was more than double that on Merueshi (1000.79), a not-quite significant difference ($F_{11,11}$=2.26, P=0.096).

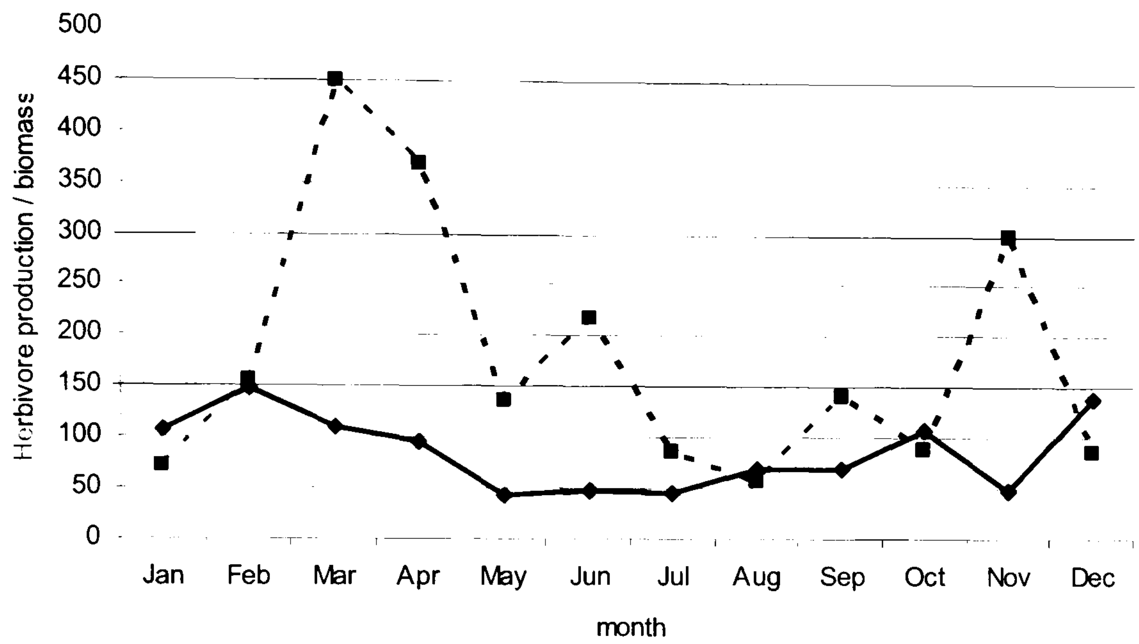
Figure 2.8 Temporal changes in biomass and ground cover, on Mbirikani and Merueshi. The solid lines represent Mbirikani; the dashed lines Merueshi.



2.3.4.3 Grazing pressure

For the purpose of investigating grazing pressure, herbivore densities were converted into 'production' in kcal/km^2/yr. Monthly ratios of herbivore production to grass biomass are shown in Figure 2.9, illustrating the relative grazing pressure on the grass resource in both ranches. Clearly, there is much greater grazing pressure on Merueshi than on Mbirikani.

Figure 2.9 Ratio of herbivore production (kcal/km^2/yr) to grass biomass (g/m^2) on Mbirikani (solid line) and Merueshi (dashed line).



2.3.4.4 Analysis of grass data by season

Tables 2.1 and 2.2 respectively show the descriptive statistics of grass in dry and wet seasons on each ranch. The results of the Mann-Whitney U-Tests are also shown. Results indicate that, with the exception of biomass in the dry season, Mbirikani had significantly greater biomass and a higher percentage ground cover by grass than Merueshi. In contrast, Merueshi grass was significantly more grazed and significantly greener than Mbirikani's grass in the wet season. The comparison of grass greenness in the dry season must be treated with caution as the data were almost entirely zero with a few positive records.

Table 2.1 Dry season descriptive statistics of four grass characteristics on Mbirikani (N=892) and Merueshi (N=594) Group Ranches, including results of the Mann-Whitney U-tests comparing the two ranches. (+ indicates Mbirikani is higher than Merueshi, whilst - indicates the opposite).

	Ranch	*Mean ±SE*	*Median (IQ range)*	*W*	*Z*	*P*
Biomass (g/m^2)	Mbirikani	116.43 ± 4.92	68.00 (40.80 - 149.50)	663938.00	-0.056	=0.955
	Merueshi	90.28 ± 3.08	68.00 (40.80 - 108.80)			
Ground Cover (%)	Mbirikani	33.83 ± 0.78	28.00 (14.00 - 58.00)	402148.00	-4.908	<0.001 +
	Merueshi	24.74 ± 0.48	23.00 (16.00 - 31.25)			
Grazing (%)	Mbirikani	58.19 ± 1.36	62.50 (21.40 - 100.00)	483487.00	-8.595	<0.001 –
	Merueshi	76.79 ± 1.45	100.00 (57.10 - 100.00)			
Green (%)	Mbirikani	3.39 ± 0.47	0.00 (0.00 - 0.00)	383074.50	-7.312	<0.001 +
	Merueshi	0.06 ± 0.04	0.00 (0.00 - 0.00)			

Table 2.2 Wet season descriptive statistics of four grass characteristics on Mbirikani (N=1596) and Merueshi (N=652) Group Ranches, including results of the Mann-Whitney U-tests comparing the two ranches. (+ indicates Mbirikani is higher than Merueshi, whilst - indicates the opposite).

	Ranch	*Mean ±SE*	*Median (IQ range)*	*W*	*Z*	*P*
Biomass (g/m^2)	Mbirikani	181.36 ± 4.61	135.90 (54.40 – 244.70)	613077.00	-8.323	<0.001 +
	Merueshi	118.90 ± 5.27	81.60 (54.40 – 135.90)			
Ground Cover (%)	Mbirikani	45.77 ± 0.63	46.00 (25.00 - 46.00)	567125.50	-11.892	<0.001 +
	Merueshi	32.04 ± 0.59	31.00 (22.00 - 40.00)			
Grazing (%)	Mbirikani	36.64 ± 0.91	29.40 (0.00 - 62.50)	1537593.00	-2.168	=0.030 –
	Merueshi	40.45 ± 1.41	40.00 (0.00 - 66.70)			
Green (%)	Mbirikani	43.17 ± 0.95	40.00 (0.00 - 80.00)	1714028.50	-2.944	=0.003 –
	Merueshi	47.83 ± 1.40	46.70 (20.00 - 75.00)			

2.3.4.5 Comparison between matched habitats - grass

Mbirikani had a greater variety of habitat types than Merueshi, although Merueshi was comprised of the two habitats that dominate Mbirikani, dense bush and thinly bushed grassland (Figures 2.3 and 2.4). A second comparison between the ranches was carried out, including only these two habitats, and only one result was different. During the wet season, the percentage greenness in matching habitat types on Mbirikani and Merueshi was not significantly different (W=424456.50, Z=-1.943, P=0.052). This shows that, even in matched habitats, Mbirikani had a higher percentage ground cover of grass, was less heavily grazed in proportion to the resource available, and had a significantly greater biomass of grass than Merueshi in the wet season.

2.3.5 Animal census

2.3.5.1 *Comparison of belt and point transects*

Statistical comparison of the two counting methods showed no evidence for a systematic bias in the density estimates derived from belt transects versus point transects. All results are given in Appendix 2B. In only two out of 72 tests (2.8%) were there significant differences between the density estimates derived from these methods. Therefore, throughout this study, results from point and line transects were considered to be directly comparable.

2.3.5.2 *Wildlife and livestock on Merueshi and Mbirikani*

Tables 2.3 and 2.4 give the descriptive statistics for wildlife and livestock densities on Mbirikani and Merueshi group ranches in dry and wet seasons respectively. The results indicate a highly significant difference in wildlife densities in both dry and wet seasons (W=69581.50, P<0.001 and W=94077.00, P<0.001 respectively), with Mbirikani having significantly higher densities [see Figure 2.10 (a)]. This significant difference applied to wild grazers, browsers and mixed-feeders independently as well. The density of livestock on the two ranches did not differ significantly in either season.

Table 2.3 Descriptive statistics of wildlife and livestock densities (in no/km^2) during the dry season for Mbirikani (N=235) and Merueshi (N=297), including results of the Mann-Whitney U-Tests for comparison between ranches. (+ indicates Mbirikani is higher than Merueshi, whilst - indicates the opposite).

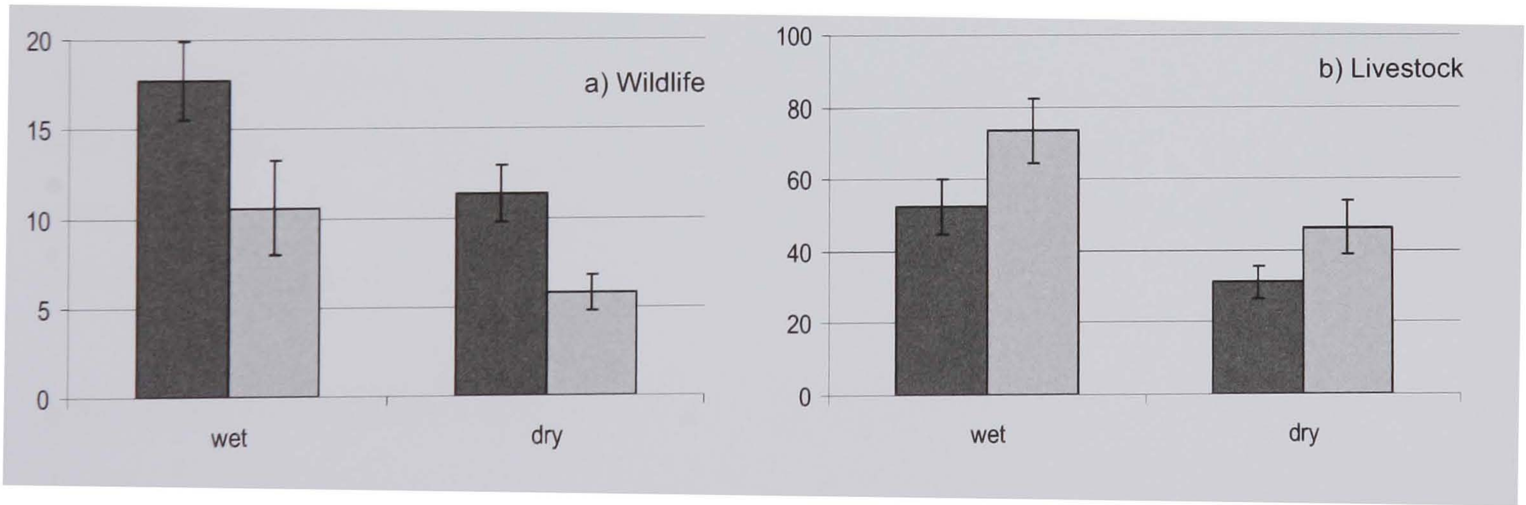
DRY SEASON	*Ranch*	*Mean ±SE*	*Median (IQ range)*	*W*	*Z*	*P*
Wild grazers	Mbirikani	7.70 ± 1.49	0.00 (0.00 - 3.57)	72354.50	-5.210	<0.001 +
	Merueshi	2.96 ± 0.68	0.00 (0.00 - 0.00)			
Wild browsers	Mbirikani	0.87 ± 0.21	0.00 (0.00 - 0.00)	73579.50	-5.761	<0.001 +
	Merueshi	0.24 ± 0.07	0.00 (0.00 - 0.00)			
Wild mixed-feeders	Mbirikani	2.79 ± 0.55	0.00 (0.00 - 1.28)	74499.50	-3.719	<0.001 +
	Merueshi	2.53 ± 0.51	0.00 (0.00 - 0.00)			
All wildlife	Mbirikani	11.36 ± 1.61	1.11 (0.00 - 10.61)	69581.50	-6.284	<0.001 +
	Merueshi	5.73 ± 1.00	0.00 (0.00 - 0.00)			
All livestock	Mbirikani	31.05 ± 4.42	0.00 (0.00 - 36.38)	62331.50	-0.202	=0.840 –
	Merueshi	46.34 ± 7.35	0.00 (0.00 - 36.78)			
All macro herbivores	Mbirikani	42.41 ± 4.80	10.83 (0.00 - 53.05)	73424.50	-3.394	=0.001 –
	Merueshi	52.08 ± 7.58	0.00 (0.00 - 45.52)			

Table 2.4 Descriptive statistics of wildlife and livestock densities (in no/km^2) during the wet season for Mbirikani (N=338) and Merueshi (N=327), including results of the Mann-Whitney U-Tests for comparison between ranches. (+ indicates Mbirikani is higher than Merueshi, whilst - indicates the opposite).

WET SEASON	*Ranch*	*Mean ±SE*	*Median (IQ range)*	*W*	*Z*	*P*
Wild grazers	Mbirikani	13.52 ± 2.03	0.00 (0.00 - 10.15)	96422.00	-6.007	<0.001 +
	Merueshi	7.99 ± 2.55	0.00 (0.00 - 0.00)			
Wild browsers	Mbirikani	1.18 ± 0.36	0.00 (0.00 - 0.00)	100294.50	-5.983	<0.001 +
	Merueshi	0.48 ± 0.16	0.00 (0.00 - 0.00)			
Wild mixed-feeders	Mbirikani	3.02 ± 0.50	0.00 (0.00 - 2.52)	98511.50	-5.527	<0.001 +
	Merueshi	2.09 ± 0.35	0.00 (0.00 - 0.00)			
All wildlife	Mbirikani	17.73 ± 2.20	2.81 (0.00 - 17.07)	94077.00	-6.608	<0.001 +
	Merueshi	10.57 ± 2.63	0.00 (0.00 - 7.07)			
All livestock	Mbirikani	52.38 ± 7.49	0.00 (0.00 - 46.44)	110468.00	-0.969	=0.332 –
	Merueshi	73.41 ± 8.93	0.00 (0.00 - 67.64)			
All macro herbivores	Mbirikani	70.11 ± 7.98	11.57 (0.00 - 75.16)	104872.00	-1.665	=0.096 –
	Merueshi	83.99 ± 9.29	7.07 (0.00 - 83.15)			

Figure 2.10 illustrates the mean seasonal densities of wildlife (a) and livestock (b) on each ranch. Wildlife densities were significantly different, whereas the livestock densities were not.

Figure 2.10 Mean densities in no/km^2 (+ standard error bars) of wildlife (a) and livestock (b), in dry and wet seasons. Dark bars represent Mbirikani, light bars Merueshi.



2.3.5.3 Comparison between matched habitats - animals

Macro-herbivore densities were compared a second time using results from only the thinly bushed grassland and dense bush on Mbirikani, in order to make a more direct comparison with the habitats on Merueshi. During the dry season, there was no difference in results obtained, with wildlife densities remaining significantly higher on Mbirikani ($W=49361.00$, $Z=-6.093$, $P<0.001$) and no significant difference in livestock densities ($W=52106.00$, $Z=-1.480$, $P=0.139$). In the wet season, the only different result was that total macro herbivore densities were significantly higher on Mbirikani ($W=66654.00$, $Z=-3.162$, $P=0.002$). Wildlife remained significantly different ($W=62652.00$, $Z=-7.647$, $P<0.001$) and livestock non-significantly different ($W=69183.00$, $Z=-0.830$, $P=0.407$) between ranches. This indicates that the differences are not simply a result of the different habitats on Mbirikani, and the results are robust even when comparing directly matched habitats.

2.4 DISCUSSION

The increased sedentarisation of pastoral populations has characterised most arid and semi-arid regions of the world during the past two millennia (McPeak & Little 2005). In Kenya, the transition to settled life, for long a primary objective of development policies aimed at pastoralists, has been forecast to cause substantial ecological and economic problems (Schwartz 2005). Numerous researchers have pointed out the potential ecological costs of this process (e.g. see Grandin, de Leeuw & Lembuya 1989; Turner 1989; Galaty 1992; Niamir-Fuller 1999). However few of these have provided detailed quantitative data, backed up with long term data, to endorse these assertions. The majority of the data presented in this chapter represents a snapshot of the ecological conditions on Mbirikani and Merueshi during 2005. However, a comparison of the two ranches would provide insufficient information on its own to draw conclusions about the effects of land subdivision. With the support of long term data however (D. Western, unpublished data), it is possible to make some assumptions about how the process of subdivision has shaped the landscape we see today.

2.4.1 Long term comparison of Mbirikani and Merueshi

Western's data, summarised in Figures 2.1 and 2.2, clearly illustrate that wildlife production on Kaputei and Mbirikani was extremely similar prior to the subdivision of the Kaputei ranches in the mid 1980s. Wildlife appeared to increase on both ranches after the drought in the early 1970s (Campbell 1999), but from the early 1990s, wildlife production showed a dramatic decline on Kaputei, whilst continuing to increase on Mbirikani. These background data allow the situation seen today to be interpreted as a divergence of once-similar wildlife production trends, rather than something that was different from the start. The same applies to the livestock production data (Figure 2.2), which provide a context for the situation seen today, and demonstrates that the differences in livestock densities between the two areas recorded in 2005 are part of a long-term, consistent trend rather than simply a one-off, potentially anomalous result.

2.4.2 Ranch characteristics

The vegetation types documented for Mbirikani and Merueshi Group Ranches are consistent with other studies. Hurt (1999) described the habitat types on Mbirikani, while Njoka (1979) studied the vegetation of Merueshi Group Ranch. Njoka (1979), using the definition of vegetation types in Pratt, Greenway & Gwynne (1966), found Merueshi to be composed of 18% dense bush (over 20% tree or bush cover) and 82% thinly bushed grassland (grassland or bushed grassland with under 20% bush cover), extremely similar proportions to those found in this study (21% dense bush and 79% thinly bushed grassland). Understanding the different habitats and physical features of each ranch provides the context for the comparison of the ranches that follows.

2.4.3 Distribution of bomas

A comparison between the distribution of bomas on Merueshi and Mbirikani (Figures 2.5 and 2.6) illustrates one of the main effects of land subdivision; a reduction in freedom of movement. In an open system, during wet months, there is usually a wide dispersal of settlements on the ranch whilst pastoral families take advantage of the widespread availability of forage and water. Settlements contract again during dry months when they concentrate around water. This is true of many semi-nomadic pastoral systems (see Schwartz 2005), and is clearly illustrated for Mbirikani Group Ranch by the distribution of dry and wet season temporary bomas (Figure 2.5). Merueshi Group Ranch however, does not demonstrate this pattern. There were almost no temporary bomas found on Merueshi Group Ranch over six different surveys. The few that were in use were mostly located off the ranch, either on Mbirikani or in the Chyulu Hills National Park. This illustrates very clearly the sedentarisation of the Merueshi community. The vast majority of people remain on their own land throughout the year, being unable to move to other parts of the ranch after water or forage due to individual land ownership. If they have to move, they are forced to either move illegally into the National Park, onto their only neighbouring ranch which remains communal (Mbirikani), or much further afield. By contrast, Mbirikani Group

Ranch members can and do make use of their ranch by moving their livestock herds out into temporary bomas, illustrating the flexibility of the system. The dependency on freedom of movement and unrestricted access to the land is emphasised by the quantity and distribution of temporary bomas on Mbirikani.

One other noted consequence of sedentarisation is that the number of households per boma decreases (Homewood 1995). This finding is supported in this study which found a mean of 1.3 households per boma on Merueshi and 2.1 per boma on Mbirikani, a highly significant difference ($W=24691.50$, $Z=-6.764$, $P<0.001$). Both this, and the number and distribution of temporary bomas is evidence of the sedentarisation of Merueshi, the consequences of which are discussed below.

2.4.4 Grass sampling

The high R^2 value of the regression equation used to transform mean blades per pin into biomass illustrates the strength of the relationship between the two variables and gives confidence in the biomass figures obtained. In addition, biomass values are consistent with those of other studies in the same region (see Mwangi & Western 1998). Results show that Mbirikani had higher grass biomass and ground cover than Merueshi in every month and these differences were significant. In addition, when comparing the ranches by season, Mbirikani still had significantly higher ground cover than Merueshi in both seasons, and a significantly higher biomass in the wet season. These results remained unchanged when comparing matched habitats, so cannot simply be attributed to habitat differences. Determining whether or not this result was an effect of land subdivision and/or sedentarisation needs careful investigation. Other factors such as different livestock stocking rates, different rainfall patterns, different soil types and different utilisation by wildlife may also be relevant.

It is difficult to completely distinguish between effects of reduced livestock mobility (i.e. sedentarisation) and differences in stocking rates (Boone 2005). My results showed no significant difference in livestock densities between the two ranches in 2005, although historically the Kaputei ranches had consistently higher livestock production than Mbirikani (D. Western, unpublished data), so lower grass biomass and ground cover may be the result of this long term trend. There is no evidence for Merueshi having lower rainfall than Mbirikani (data from this study), and in fact there is a north-south rainfall gradient in this part of Kenya and Merueshi usually receives slightly *more* rain than Mbirikani (de Leeuw

1991). Soil types on the ranches were not very different (de Leeuw 1991), making the differences in grass biomass unlikely to be as a result of soil-type.

Higher densities of wildlife may be a cause of reduced grass availability. However, this study found that Merueshi had significantly less wildlife than Mbirikani (Tables 2.3 and 2.4), and long term data shows that this had been the case since the early 1990s (see Figure 2.1). A study by de Leeuw (1991) found that in the early 1980s, i.e. before subdivision was underway, Mbirikani had a lower standing biomass of grass than Merueshi, which suggests that the opposite situation observed today may in fact be a result of subdivision.

Figure 2.9 shows there was consistently less grass per unit of animal production on Merueshi as compared with Mbirikani, i.e. the rangeland was more heavily pressured throughout the year. Further evidence of this is provided by the significantly greater percentage of grazed grass on Merueshi (Tables 2.1 and 2.2), and the reduced seasonal variation of grass biomass on Merueshi (Figure 2.8). These results illustrate the effects of sedentarisation, where grazing pressure cannot be rotated and rangelands are subjected to continual pressure.

In summary, the lower grass biomass and ground cover recorded on Merueshi may be partly due to higher livestock densities, but results mostly from the reduced mobility caused by land subdivision, thus supporting the sub-hypothesis of Section 2.3.4. This finding is supported by theoretical investigation of land fragmentation. Boone (2005) modelled the effects of subdivision on vegetation in South Africa. His results showed significant declines in herbaceous biomass as a 300km^2 landscape was divided into smaller and smaller parcels. High palatability grasses and their root stock declined most rapidly, while low palatability grasses increased slightly. There was a progressive decline in annual net primary productivity and an increase in trees and shrubs as parcel sizes shrank and livestock were forced to graze in the same patches all year (Boone 2005).

In addition, Njoka (1979) showed a marked downward trend in range condition in the south Kaputei ranches (including Merueshi) between 1969 and 1977, with desirable grasses decreasing by 75% during that time period. That marked the first nine years after the demarcation of the Kaputei Group Ranches in 1968 (Kimani & Pickard 1998), and indicates that land adjudication, even into relatively large parcels like the group ranches (mean of about 150km^2) can cause rangeland deterioration. However, there was a substantial drought during that time period (Campbell 1999), so the deterioration of the rangeland cannot be attributed solely to land adjudication.

2.4.5 Animal census

The use of two different techniques to measure the same thing (animal densities) can only be justified if there is no inherent bias within the techniques. I found no significant differences in results obtained from point and belt transects. This is an important finding, since there are likely to be many occasions in scientific fieldwork where the use of a combination of these techniques is necessary. In this study, it allows the results from each method to be treated and interpreted in exactly the same way.

2.4.5.1 Wildlife

Monthly population census results showed that in 2005, in both dry and wet seasons, Mbirikani supported a significantly higher density of wildlife than Merueshi, even in matched habitats. This finding is supported by the aerial counts from the same period which showed wildlife production (a function of density and weight) to be higher on Mbirikani than Kaputei. A reduction in wildlife populations is one of the recognised consequences of land subdivision (Seno & Shaw 2002). However a one-off snapshot of wildlife populations in two neighbouring ranches does not illustrate a decline in wildlife, but only that in 2005, wildlife populations were significantly lower on Merueshi than Mbirikani. However, the data summarised in Figure 2.1 do provide evidence of the decrease in wildlife since 1990. Nonetheless, this is still insufficient evidence on its own to infer that land subdivision and sedentarisation are the root cause. Other factors which may be involved include increasing livestock populations, a reduction in the availability of water on the ranch, a reduction in availability of forage and an obstruction of the wildlife migration routes.

I believe that it is a combination of these reasons which has resulted in the dramatic decline of wildlife on Merueshi and the surrounding Kaputei ranches in past 10-15 years. However, I propose that each reason, with the exception of increases in livestock, is in itself caused by land subdivision. A reduction of available surface water would be sufficient to cause a decline in wildlife populations (Western 1975). In the case of Merueshi Group Ranch, once the land was subdivided, all water access points were fenced off so that the owner could have priority access for his household and livestock, and charge other people to water their livestock there (Merueshi members, *pers. comm.*). This fencing of previously communal resources is only possible once land has been subdivided. The situation in 2005 was that the only water available to wildlife was from the borehole at Merueshi town which was so heavily utilised by livestock that use by wildlife would have been restricted to night times.

A reduction in the availability of forage would also result in a decrease in wildlife, especially grazers as it is the grass resource which is most vulnerable to change. Indeed, although densities of grazers, browsers and mixed feeders were all significantly higher on Mbirikani than Merueshi, the greatest inter-ranch difference was in the density of wild grazers (Tables 2.3 and 2.4). As discussed previously, the fact that Merueshi has significantly less grass than Mbirikani is most likely a result of land subdivision. Additionally, the removal of thorny trees and shrubs to create the fences to demarcate private property (Smit 2004; Kolowski & Holecamp 2006) might result in a decrease in the quantity of browse available. This in turn may be partly responsible for the lower density of browsers witnessed on Merueshi. However, it is well established that heavy livestock grazing causes bush encroachment (see Hudak 1999; Moleele *et al.* 2002) and indeed on Merueshi there does not appear to be a shortage of woody vegetation. The decrease in browsers may therefore be mostly due to reduced water availability and increased human disturbances.

A blocking of wildlife migration routes is considered one of the most detrimental consequences of land subdivision (Odundo 1992; World Bank 1994). Indeed initially, fencing of the original group ranches in Kajiado District was prohibited, since they incorporated the main game migration routes between the Serengeti, Amboseli and Tsavo National Parks (Graham 1988). However, once these ranches were subdivided and under individual land ownership, there was nothing to stop people fencing their land and the result is a mosaic of fenced and unfenced plots, entirely unplanned and with no central management. Such *ad hoc* fencing can exert harmful effects on ecosystems and may truncate migratory movements of wildlife (Hailey & DeArment 1969; Boone & Hobbs 2004). In addition livestock or wildlife confined to or excluded from land parcels may overgraze the available vegetation (Boone & Hobbs 2004), leading to land degradation and an ultimate decline in population numbers. Both have occurred on Merueshi Group Ranch.

Findings from this study therefore agree to an extent with Stanley's statement that wildlife 'disappears' in subdivided areas (Stanley 2000). However, as mentioned, the other factor which may also play a part in this decrease of wildlife is the increase in livestock, since livestock production has increased steadily on Kaputei since the mid 1970s (Figure 2.2). Nevertheless, livestock production also increased at a very similar rate on Mbirikani during the same time period, and there was no corresponding decrease in wildlife on Mbirikani.

In summary therefore, land subdivision resulted in an obstruction of wildlife migration routes, a decrease in the quality and quantity of grass and reduced the availability of water for wildlife. Through a combination of these factors, subdivision has caused a decrease in wildlife populations on Merueshi to the extent that today, wildlife densities there are significantly lower than its neighbouring communal ranch, Mbirikani.

2.4.5.2 Livestock

Relative livestock holdings have not changed all that much over the past three decades. Western's unpublished data show that livestock production on Kaputei has been greater than on Mbirikani since the 1970s. This study found that Merueshi had a higher density of livestock than Mbirikani [Figure 2.10 (b)], although not significantly so. This finding appears contrary to some author's predictions that subdivision should cause a decline in livestock holdings. For example Boone *et al.* (2005), using the ecosystem model SAVANNA, predicted that in Eselengei Group Ranch (which neighbours both Mbirikani and Merueshi), livestock carrying capacity would decline by 25% when subdivided into 1km^2 parcels. However, Merueshi Group Ranch, which subdivided early when membership was still low, was divided into parcels almost twice this size (450 acres or 1.8km^2) and seems to have escaped a sizeable decrease in livestock holdings. Nonetheless, in an informal survey (N=81), 64% of Merueshi members still reported a decrease in livestock holdings since subdivision.

The planned subdivision on Mbirikani entails even smaller land parcels than in the Eselengei model (Boone *et al.* 2005), with around 5000 plots of 60 acres (0.24km^2) to be demarcated (Mbirikani Group Ranch committee 2007, *pers. comm.*). This is more than seven times smaller than the plots on Merueshi, and would almost certainly result in a decline in livestock holdings (and an even more severe decline in wildlife densities), especially if people were to remain restricted to their individual plots. However, this is an unlikely scenario, as the Maasai realise the importance of freedom of movement for their livestock and would almost certainly set up grazing associations and try to retain more extensive use of the land (Boone *et al.* 2005; Mwathi *et al.* 2005).

2.4.6 Conclusion

Pressures to allocate land to individuals rather than groups were both internal and external. Group ranch members were seeking title to parcels as a means of retaining land, i.e. for reasons of security of tenure (99%), and due to impatience with the group ranch leadership (Boone *et al.* 2005). In addition, the Kenyan government was seeking privatisation of land as initial steps towards development (Graham 1988).

Whilst land allocation is therefore inevitable (Boone *et al.* 2005), results from this and many other studies suggest that further, physical subdivision of the rangelands is best avoided in order to maintain the integrity of the ecosystem. The challenge is to find a compromise solution that provides people with security of tenure, whilst avoiding the subdivision of the land and the consequent ecological degradation witnessed on Merueshi.

There are a number of different ways in which group ranches can be subdivided. The most straightforward is to divide the land based on the ratio of group ranch lands to the number of members, which is what occurred on Merueshi Group Ranch in the 1980s. More recently however, as communities and conservationists alike gain a deeper understanding of the problems with such a method of subdivision, it is more common to see arrangements where members receive small parcels for permanent settlement, but core areas remain open to communal grazing (Boone *et al.* 2005), and in some cases land is also left aside as a conservation area. An example of the latter is found in Siana Group Ranch near the Maasai Mara Reserve where an area of not less than 30,000 acres is being maintained under group title as a conservation area, whilst the rest of the ranch undergoes subdivision (Mwathi *et al.* 2005). A similar scenario would probably be the best solution for Mbirikani as well, and with proper management and implementation could maintain the ecosystem integrity whilst fulfilling the needs of the community. One alternative to land subdivision is explored and discussed in Chapter 7; this chapter simply serves to highlight the importance of such alternative solutions.

This chapter has shown how land subdivision can result in a decrease in wildlife numbers, especially grazers. The following chapter explores whether subdivision and sedentarisation also affects the distribution patterns of the remaining wild grazers.

CHAPTER 3

THE EFFECTS OF LAND SUBDIVISION ON THE DISTRIBUTION PATTERNS OF WILD GRAZERS

Hypothesis: Land subdivision and sedentarisation of Maasai pastoralists compromises the ability of wild grazers to distribute themselves optimally within the landscape.

ABSTRACT

Understanding the spatial dynamics of landscape use by free-ranging herbivores is crucial for ecosystem management. In this study, binary logistic regression analyses were used to determine the factors influencing the distribution of wild grazers on two Maasai ranches in Kenya's Amboseli-Tsavo Ecosystem. The main aim was to investigate whether or not land use and/or the sedentary nature of pastoralists affected wild grazer distribution patterns. I found very little evidence that grazer distributions were affected by land subdivision or sedentarisation of pastoralists. Instead I found that grazers consistently located themselves where grass biomass was highest, with grass *quality* having little effect, suggesting that forage quantity may be the limiting factor where grass biomass is generally low. The availability of surface water did not appear to significantly influence the likelihood of finding grazers present, even in the dry season. Suggestions for widening the criteria in the analyses and other suggestions for improvement are discussed.

3.1 INTRODUCTION

3.1.1 Background

According to the World Conservation Monitoring Centre (1992), the Somali-Maasai region of East Africa represents what is probably the world's richest grassland zone. Roaming within this landscape are a great variety of herbivores of different sizes, feeding strategies, gut morphologies and behaviour. In Kenya the vast majority (>70%) of wildlife lives on community lands (Grunblatt *et al.* 1995b) and is therefore vulnerable to human-induced changes regarding land use. Results from Chapter 2 illustrate that land subdivision can lead to a considerable decline in wildlife populations. In this chapter, I investigate whether subdivision also affects the distribution patterns of the remaining herbivores, and whether it compromises their ability to locate themselves optimally within the local landscape.

Countless models have been applied to describing and predicting herbivore movement patterns at every scale. Relatively few optimal foraging theory studies have focussed on herbivores, primarily because of the difficulty in defining discreet food items and complications imposed by digestive constraints (Bailey *et al.* 1996), but those that have, have been only moderately successful (Owen-Smith 1979; Owen-Smith & Novellie 1982). Ideal free distribution (IFD) models have been commonly used to try and predict animal distributions (e.g. Harper 1982; Schilling 2005), although rarely of macro-herbivores (see Kennedy & Gray 1993). Indeed most attempts to use IFD for foragers across wide spatial areas have been unsuccessful (Tyler & Hargrove 1997). This is most likely due to violation of two of the main assumptions of IFD models, specifically that individuals are free to move to any patch and that movement costs are negligible (Tyler & Hargrove 1997). Fryxell (1991) applied mechanistic optimal foraging models to try and understand aggregations of large herbivores, and Wilmshurst, Fryxell & Bergman (2000) derived a model to predict variations in habitat selection by herbivores of different sizes.

None of these models are applicable to this study however, either due to violations of assumptions, or because I am investigating overall population distribution patterns at a landscape scale, rather than foraging choice by individuals. In addition, my aim is not to *predict* future distribution patterns, but simply to investigate the causal factors behind observed distribution patterns. Binary logistic regression analyses (Redfern *et al.* 2003) and multiple regression analyses have been successfully used in other studies to explain grazing distribution patterns (Senft, Rittenhouse & Woodmansee 1983; Gillen, Krueger & Miller 1984). I therefore use logistic regression and multiple linear regression models to investigate to what extent the people, livestock and land use are having an impact on the distribution of wild grazers, and whether or not grazers are able to select areas on the basis of forage quality and quantity. Specifically, I aim to investigate the following hypothesis:

"Land subdivision and sedentarisation of Maasai pastoralists compromises the ability of wild grazers to distribute themselves optimally within the landscape."

To do this, I compare two neighbouring Maasai group ranches with similar landscape heterogeneity but different land use policies and openness of the landscape. (See

Chapter 1 for a detailed description of the study area). On Merueshi, the fragmented landscape may have compromised ideal herbivore distribution patterns, while on Mbirikani wildlife had virtually unrestricted access to all resources.

Many landscape-scale models of herbivore distributions focus primarily on the role of biotic factors such as forage quality and quantity (Redfern *et al.* 2003). However, Bailey *et al.* (1996) suggest that abiotic factors, such as slope and distance to water, are equally as important as biotic factors, and act as the primary determinants of large scale distribution patterns. Redfern *et al.* (2003) suggest that a combination of these two types of factors may be particularly important in determining the distribution patterns of large herbivores in African savannah ecosystems. Theoretically, in an environment entirely free of constraints, one would expect herbivores to locate themselves where they are able to maximise their energy gain in the shortest possible time (see Bergman *et al.* 2001). For example, Bailey *et al.* (1996) suggest that large herbivores should spend most time in areas where the there is the highest available quantity and quality of forage.

However, where the land is shared by humans and their livestock, it is possible that human activities may interfere with animal distributions, and pre-empt access by wildlife to critical habitats (Corfield 1973; Williamson, Williamson & Ngwamotsoko 1988). Additionally, there may be natural restrictions on distributions, such as the availability of water, competitive interactions with other wildlife or livestock and the effects of predation (Sinclair 1985; Fryxell 1991). In this study, constraints due to water availability and effects of competitive interaction with livestock are investigated alongside the influence of biotic factors such as grass quality and quantity. Predator densities are generally low on both ranches so the effects of predation are taken to be minimal and are not included in the analyses. In summary, this chapter investigates the relative influence of biotic, abiotic and human factors in determining the landscape-scale distribution patterns of wild grazers in the Amboseli-Tsavo Ecosystem in southern Kenya.

3.1.2 Research objectives

To investigate whether or not land subdivision influences the distribution patterns of wild grazers, the following are the specific objectives of this chapter.

- To describe and quantify the difference in landscape heterogeneity between Mbirikani and Merueshi Group Ranches.

- To investigate whether or not wild grazers display a non-random distribution pattern with a tendency to congregate in certain areas.
- To determine which factors best explain the observed distribution of wild grazers on Mbirikani and Merueshi Group Ranches, using logistic regressions.

3.2 METHODS

Logistic regression analyses (using presence/absence) were used to investigate which factors were most important in affecting the distribution pattern of wild grazers on Mbirikani and Merueshi Group Ranches. Species included in the wild grazer category were zebra, wildebeest, Thomson's gazelles, Coke's hartebeest and oryx, which were the only true grazers in the region.

Density estimates of wild grazers were obtained from monthly population counts using belt and point transects. On Mbirikani Group Ranch, 22 randomly located belt transects of 4km in length were driven each month, whilst on Merueshi Group Ranch 50 point transect counts were done monthly. The density estimates derived from these two methods were found to be directly comparable (Chapter 2). For each transect, all animals of Thomson's gazelle size or above within the specified area were counted (details in Chapter 2). The number of individuals of each species of grazer counted in a transect were summed and the result divided by the area of the transect to give a density of wild grazers per transect.

The density of wild grazers per transect, coded as 1 for presence (density >0) and 0 for absence, was used as the dependent variable. There were initially 10 possible independent variables which could have been included in the models. Three however (woody vegetation density, boma density and ground cover by grass), were excluded because of collinearity issues with each other and with grass biomass. Collinearity was determined by use of collinearity statistics (VIF and tolerance scores) and collinearity diagnostics (eigenvalues and variance proportions). It was decided to exclude these three variables, rather than any others, for the following reasons. Density of woody vegetation is less relevant than the other variables to the hypothesis being investigated, the human influence represented by 'boma density' could be investigated using another variable 'distance to nearest permanent boma' which also supplied more detail, and ground cover was significantly positively correlated with biomass in all scenarios, so biomass could be used to represent effectively the 'quantity of grass available'. The seven independent variables retained in the models are given in Table 3.1

Table 3.1 The independent variables used in the binary logistic regression analyses investigating grazer distribution patterns.

	Variable	*Description*
Human variables	Livestock	continuous variable; density of livestock (no/km^2)
	Distance to boma	continuous variable; distance to nearest permanent boma (m)
Biotic variables	Biomass	continuous variable; above-ground grass biomass (g/m^2)
	%CP	continuous variable; crude protein content of grass (%)
	%DOM	continuous variable; dry organic matter digestibility of grass (%)
	Greenness	continuous variable; percentage greenness of grass (%)
Abiotic variables	Distance to water	continuous variable; distance to nearest surface water (m)

Four biotic, grass-related variables were included in the model because past research suggests that these are frequently the most important factors determining the distribution of wild grazers within a landscape (Fryxell 1991; Wilmshurst *et al.* 2000). The most important abiotic factors are considered to be water availability and slope (Bailey *et al.* 1996). However, the majority of the landscape considered in this analysis was flat, so the only abiotic variable considered was distance to water. An investigation of the effects of human influences on the distribution of grazers was of particular relevance to the hypothesis that subdivision constrains optimal grazer distribution, hence the inclusion of livestock density and distance to nearest permanent boma as predictors.

3.2.1 Data collection methods

These independent variables were available at two different resolutions. Many variables were at the same resolution as the dependant variable (density of wild grazers), i.e. at the transect level. These included livestock densities, grass greenness, grass biomass, distance to water and distance to the nearest permanent boma. The other independent variables, %CP and %DOM, were at a coarser resolution as they were collected separately from the transects.

During each belt and point transect, grass measurements were taken, either every 500m along the belt transects or twice within the area counted by each point transect. At each sampling point, a pin-frame was used to record grass biomass (once calibrated) and percentage grass greenness (Mwangi & Western 1998). The step-point method was used to measure ground cover (Sutherland 1996). Full details of these methods are given in Chapter 2. The nine grass measurements per belt transect and two per point transect were averaged to give an estimate of grass greenness, biomass and ground cover per transect. For grass biomass on Mbirikani however, although it was collected at the transect level, it was not recorded for *every* transect, so results were averaged by habitat and included as a habitat level variable to avoid having to reduce the sample size at the transect level.

The position of all available surface water was recorded once in both dry and wet seasons using a GPS. Water points were located by Maasai assistants who knew their local areas intimately. The GPS position of every transect was also recorded; for the belt transects, the middle point of the line was used. The nearest-features extension in ArcView GIS (v3.2) was used to calculate the distance of each transect to the nearest surface water. The same technique was applied to calculate distance to the nearest permanent boma, the GPS positions of which had all been recorded in surveys carried out in 2005. The location of permanent bomas was the same for dry and wet seasons, although water availability differed considerably by season. All water points known on the ranch were included as 'available water' for the wet season months, whereas for dry season months, only permanent surface water was considered available (i.e. the swamps, river, water pipeline and safari lodge water hole).

For %CP and %DOM, data were produced for each of the seven major habitats on Mbirikani Group Ranch; boulder field, dense bush, thinly bushed grassland, drained grassland, poorly drained grassland, woodland and whistling thorn scrub (see maps in Chapter 2), or for five areas defined by Dirichlet tessellations for Merueshi. Grass forage quality (%CP and %DOM) was determined by near-infrared spectroscopy of cattle faeces (Lyons & Stuth 1992; Fahey & Hussein 1999). Fresh cattle faecal samples were collected from bomas distributed across both ranches on the same day of every month. Samples were air dried in a solar-drier in the field, then analysed by the Kenya Agricultural Research Institute (KARI) using a Near Infrared Spectrophotometry (NIRS) machine. After being oven dried at 60°C for 24 hours, the samples were ground with a cyclone grinder to produce cuboids of uniform size. Ground samples were placed in coin-envelopes, labelled and oven dried again at 60°C for 24 hours. Samples were then packed into 3g sample caps and returned to the oven in a dessicator for 3-4 hours to absorb all moisture. The spectra of the samples was then read in the NIRS machine and results analysed with help from Dr Robert Kaitho (Texas A&M University).

All data were split into dry and wet seasons for analysis, because different variables may be important in affecting the distribution of grazers in the different seasons. This was done on the basis of grass greenness and percentage biomass deviations from the overall biomass mean (Mose 2005). This method is described in detail in Chapter 2. Overall, this produces four different scenarios to be analysed independently; Mbirikani wet and dry season distributions, and Merueshi wet and dry season distributions.

3.2.2 Binary logistic regressions

Logistic regressions were carried out in SPSS v12.0 using the enter method. This method was chosen above the stepwise methods (forwards, backwards and stepwise) because there is sound theoretical literature available to base the model on, precluding the need to rely on the computer to select variables based on mathematical criteria (Field 2006). In addition, this method consistently produced the best model with regard to several different criteria; -2 log-likelihood statistics, the Nagelkerke R^2 values, the Hosmer-Lemeshow goodness of fit tests and the classification plots produced in SPSS.

Having selected the enter method for the reasons given, use of Akaike's Information Criterion (AIC) in model selection was investigated. AIC scores and Delta AIC were calculated for all possible models by removing the least significant variable each time and re-running the model. However, ultimately the global model is presented in all cases. The global model is defined as the most complex model of the set, i.e. the one which includes all the variables of interest (Mazerolle 2006). This is because, for all scenarios, the global model was close to the 'best' model on the basis of the lowest AIC score, and significant variables were so strong that they remained consistent in all the different models. In addition, the global model provides most information by including information on the direction and strength of non-significant variables, which can be just as interesting as significant variables (Field 2006).

The case summaries of the final models (including Cook's distance values, Leverage values, standardized residuals, and DFBeta values) were examined to ensure there were no individual points which were having an especially strong influence on the model. If statistical outliers were discovered, the data were examined to see why the point was having an unusually high influence. Raw data sheets were also examined to ensure there were no errors in the original data input. Treatment of individual outliers is discussed in the relevant section of the results.

3.2.3 Multiple linear regressions

A second stage of analysis was carried out on the data using multiple linear regressions, including only cases where wild grazers were present. This method was not possible initially due to the large numbers of zeros in the dataset which violated assumptions of linear regression analyses. However, once all transects with an observation of zero wild grazers were excluded from the dataset, and the dependent variable log transformed, it was possible to carry out multiple linear regressions. Results from this stage of the sampling showed what affected the distribution densities of wild grazers given that they were present. The same independent variables were used in both stages. Results of these analyses did not add much to the results obtained from the binary logistic regressions, and these are therefore given in Appendix 3A.

3.3 RESULTS

3.3.1 Comparison of landscape heterogeneity

A comparison of factors affecting the distribution of wild grazers would be fairer if the two areas to be compared do not show great differences in landscape heterogeneity. An F-test was used to compare the variances of grass biomass on Mbirikani and Merueshi (10925.30 and 11139.48 respectively). Values used were the means of results from each transect, in order to avoid bias from a few extreme sample points. The test showed no significant difference between the ranches ($F_{697,622}$=1.02, P=0.400). To investigate differences in woody vegetation density on the two ranches, a Monte-Carlo re-sampling technique (100,000 samples with replacement) was used, in order to account for sample size differences (N=63 and N=12 for Mbirikani and Merueshi respectively). Results showed a non significant difference between the ranches (95% CI for Mbirikani variances = 0.265 to 2.782, Merueshi variance = 2.74). Since neither measure of heterogeneity differed significantly between the ranches, direct comparisons can be made of factors affecting grazer distributions.

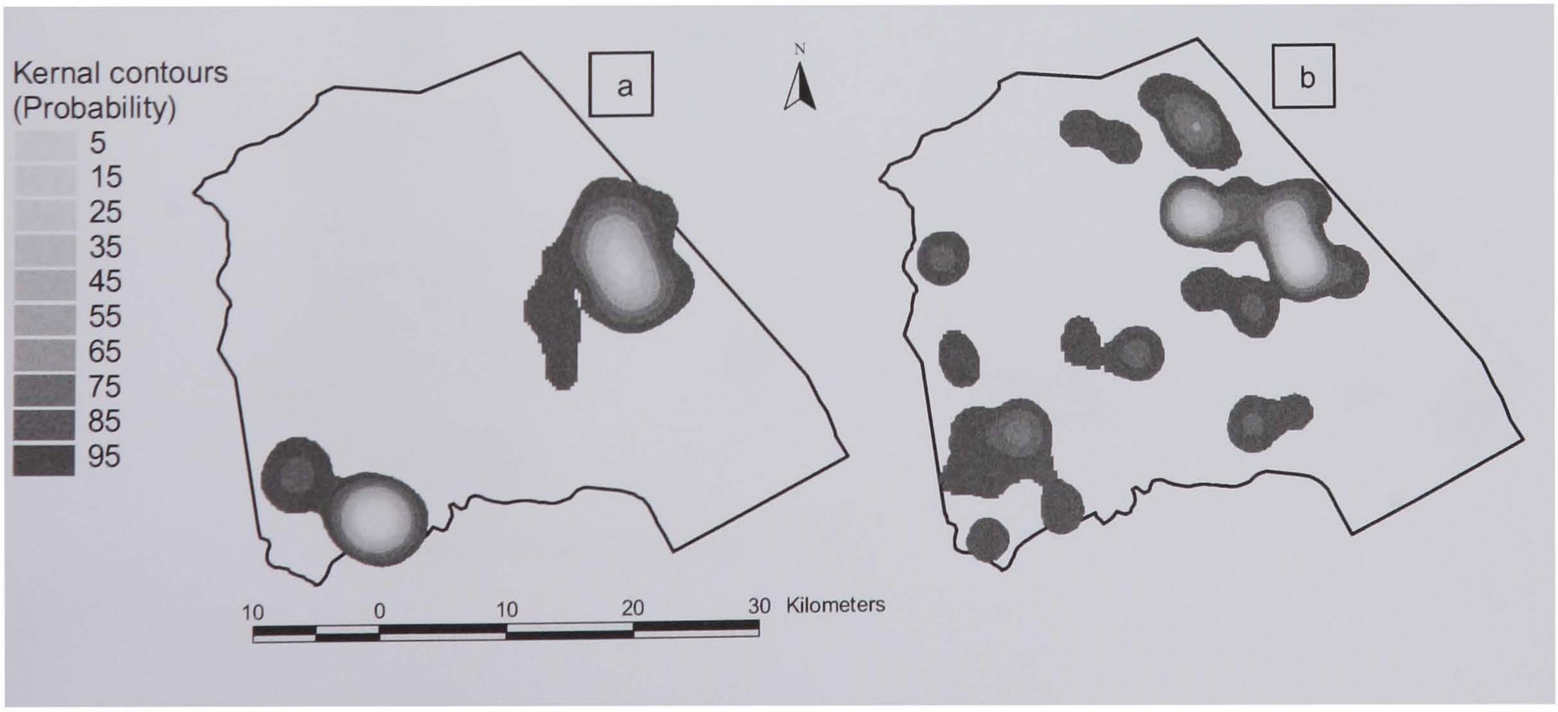
3.3.2 Distribution of wildlife on Mbirikani Group Ranch

The distribution of wildlife between different habitats on Mbirikani Group Ranch was significantly different from what would be expected if the animals distributed themselves evenly with respect to the size of the habitat (for a map of the habitats, see Figure 2.3 in Chapter 2). This applies to both dry and wet seasons (χ^2_6=7898.55, P<0.001 and χ^2_6=7787.17, P<0.001 respectively). In general, the greatest densities of wildlife were found in the smallest habitats (woodland and grasslands), suggesting a particular preference for those habitats (see Table 3.2). This is illustrated in Figure 3.1.

Table 3.2 Observed densities (in no/km^2) of wild grazers by habitat on Mbirikani Group Ranch in dry and wet seasons. % = the percentage of the total wild grazer population found in that habitat.

Habitat	*Area (km^2)*	*Dry season densities*	*% (dry)*	*Wet season densities*	*% (wet)*
Woodland	23.7	18.04	23	17.15	11
Drained grassland	39.7	31.72	40	39.79	27
Poorly drained grassland	70.2	3.92	5	45.90	31
Whistling thorn scrub	97.4	13.28	17	17.66	12
Boulder field	170.5	5.49	7	4.46	3
Thinly bushed grassland	225.7	5.62	7	12.64	8
Dense bush	444.0	1.36	2	12.18	8
TOTAL	**1071.20**	**79.43**	**100**	**149.77**	**100**

Figure 3.1 Maps (using Kernal home ranges) to illustrate the distribution of wild grazers on Mbirikani Group Ranch in the dry season (map a) and the wet season (map b). Concentrations of grazers are indicated by the paler tones. See Figure 2.3 (Chapter 2) for underlying habitats.

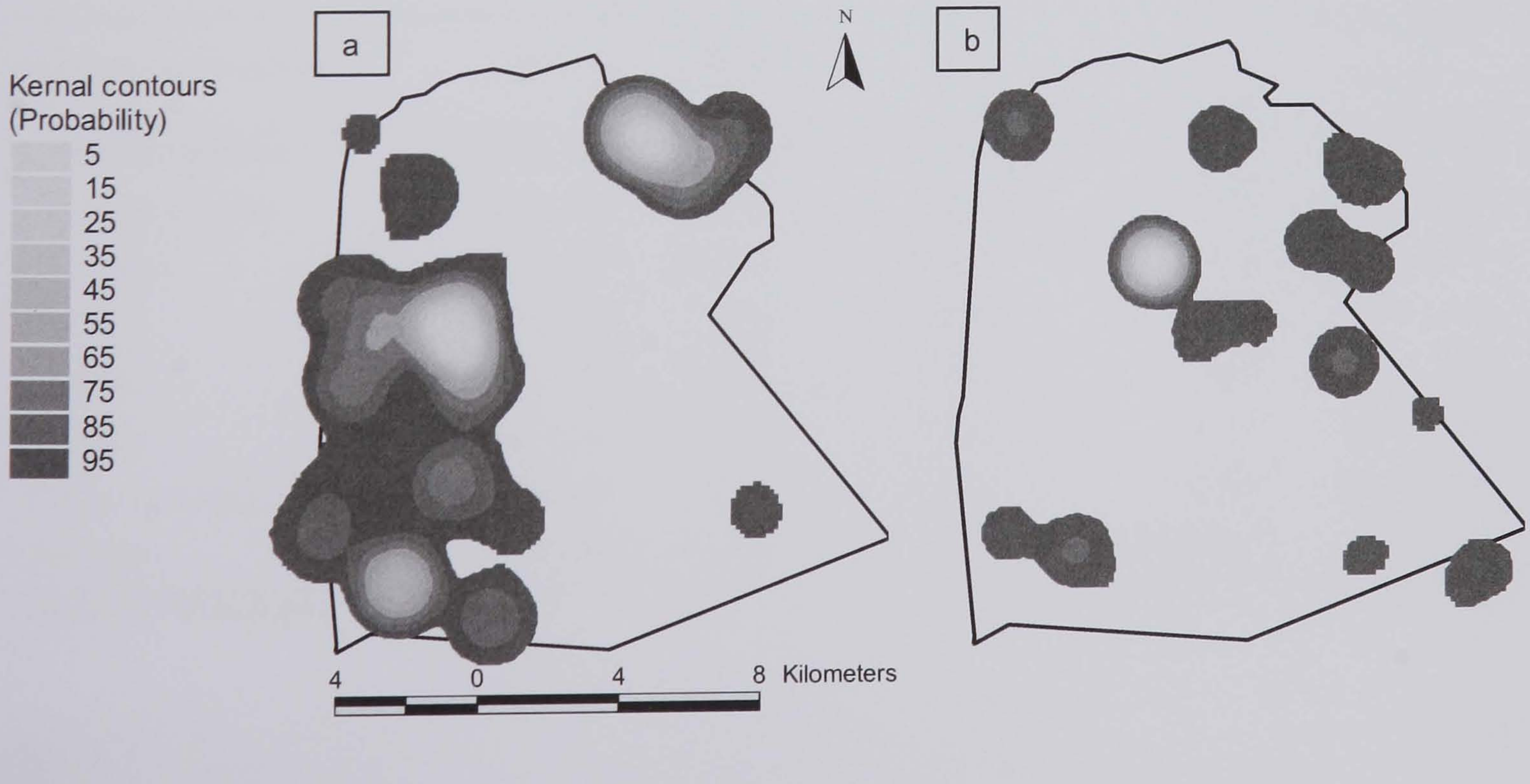


Regression analyses were used to investigate what factors may be involved in making the wild grazers congregate in these areas. Were they selecting optimal grazing resources or was their aggregation in these areas due to displacement by people or livestock? This was investigated by logistic regression analyses, the results of which are given in section 3.3.4 and 3.3.5.

3.3.3 Distribution of wildlife on Merueshi Group Ranch

There were no separate habitats defined on Merueshi due to its more patchy nature and lack of large, distinct habitat types. Therefore, any patterns within the distribution of wild grazers on Merueshi were tested using an index of dispersion against a null hypothesis that animals distributed themselves randomly on the ranch. Merueshi Group Ranch was divided into 65 grids using ArcView v3.2. The numbers of individual wild grazers in each grid were counted, as well as the number of point transects located in each grid. Wildlife densities were standardized for sampling effort by dividing by the number of transects done in each grid. The results showed that the distribution of wildlife was significantly different from random (χ^2_{64}=655.657, P<0.001) and could be classified as clumped (variance/mean ratio = 10.252) (Fowler *et al.* 1998). These results are the same for wet and dry seasons independently. Grazer concentrations are illustrated in Figure 3.2.

Figure 3.2 Maps (using Kernal home ranges) to illustrate the distribution of wild grazers on Merueshi Group Ranch in the dry season (map a) and the wet season (map b). Concentrations of grazers are indicted by the paler tones.



Wild grazers were clearly not randomly distributed over either ranch, instead showing a highly clumped distribution pattern. This result provides the basis for further investigation of which factors may be involved in causing the observed distributions.

3.3.4 Wet season regression results

This section presents results of the binary logistic regression models for Mbirikani and Merueshi wet season grazer distributions. In each case, the variables revealed as significant in the global model presented were consistently found to be significant in a preliminary investigation of the data, so the results are felt to be robust and accurate.

3.3.4.1 Mbirikani wet season - binary logistic regression results

The model produced a significant result overall (χ^2_7=106.803, P<0.001) and had a good fit (Hosmer & Lemeshow; χ^2_8=6.600, P=0.580; Nagelkerke R^2=0.368). Three variables emerged as significant in explaining the likelihood of finding grazers present. Both a greater biomass and a greater percentage greenness of the grass significantly increased the odds of grazer presence. A higher %DOM significantly decreased the odds of grazer presence. All results of the model are shown in Table 3.3.

Table 3.3 Results of the binary logistic regression model for investigating presence/absence of grazers on Mbirikani in the wet season (N=333).

Independent variables	*B ± S.E*	*Wald*	*Sig.*		*Exp(B)*	*95% C.I.s for Exp(B)*
Livestock densities	.000 ± .001	.013	0.910		1.000	.998 – 1.002
Distance to boma	.000 ± .000	.269	0.604		1.000	1.000 – 1.000
Biomass	.014 ± .002	32.798	0.000	***	1.014	1.009 – 1.019
%CP	-.419 ± .291	2.068	0.150		.658	.372 – 1.164
%DOM	-.156 ± .072	4.693	0.030	*	.855	.743 – .985
% green	.015 ± .004	14.908	0.000	***	1.015	1.007 – 1.022
Distance to water	.000 ± .000	2.942	0.086		1.000	1.000 – 1.000
Constant	10.552 ± 4.483	5.540	0.019		38246.641	

*=P<0.05, ***=P<0.001

3.3.4.2 Merueshi wet season - binary logistic regression results

Cook's distance influence statistics and an anomalously high standardized residual for one case in this dataset suggested an outlier. The case was examined and found to have a very high biomass value. The raw data were checked to ensure there was no error in the data input. Even so, the point was excluded from the model because it was a statistical outlier and biologically non-representative of the vast majority of the known biomass values. Nonetheless, the overall outcome of the model was the same with and without this point.

The final model produced a highly significant result (χ^2_7=27.796, P<0.001) and had a good fit (Hosmer & Lemeshow; χ^2_8=4.267, P=0.832; Nagelkerke R^2=0.129). Only one variable was found to have a significant effect on the distribution of wild grazers on Merueshi in the wet season; a higher grass biomass significantly increased the odds of grazer presence. All model results are shown in Table 3.4.

Table 3.4 Results of the binary logistic regression model for investigating presence/absence of grazers on Merueshi in the wet season (N=320).

Independent variables	*B ± S.E*	*Wald*	*Sig.*		*Exp(B)*	*95% C.I.s for Exp(B)*
Livestock densities	.000 ± 001	.233	.629		1.000	.999 – 1.002
Distance to boma	.000 ± .000	.548	.459		1.000	1.000 – 1.001
Biomass	.005 ± .001	17.121	.000	***	1.005	1.003 – 1.008
%CP	-.090 ± .100	.798	.372		.914	.751 – 1.113
%DOM	-.245 ± .202	1.461	.227		.783	.527 – 1.164
% green	.002 ± .005	.271	.603		1.002	.993 – 1.011
Distance to water	.000 ± .000	.337	.561		1.000	1.000 – 1.001
Constant	12.611± 12.057	1.094	.296		299946.855	

***=P<0.001

3.3.5 Dry season regression results

3.3.5.1 Mbirikani dry season - binary logistic regression results

For this analysis, %CP had to be removed because of collinearity issues with both biomass and %DOM. The model was highly significant (χ^2_6=91.338, P<0.001) and had a good fit (Hosmer & Lemeshow; χ^2_8=25.009, P=0.002; Nagelkerke R^2=0.445). Only biomass had a significant relationship with the probability of finding wild grazers present. Results are shown in Table 3.5.

Table 3.5 Results of the binary logistic regression model for investigating presence/absence of grazers on Mbirikani in the dry season (N=235).

Independent variables	*B ± S.E*	*Wald*	*Sig.*		*Exp(B)*	*95% C.I.s for Exp(B)*
Livestock densities	.001 ± .003	.065	.799		1.001	
Distance to boma	.000 ± .000	.038	.845		1.000	1.000 – 1.000
Biomass	.025 ± .005	24.675	.000	***	1.025	1.015 – 1.035
%DOM	.028 ± .113	.060	.807		1.028	.824 – 1.283
% green	.064 ± .049	1.708	.191		1.066	.969 – 1.173
Distance to water	.000 ± .000	.261	.609		1.000	1.000 – 1.000
Constant	-4.943 ± 6.665	.550	.458		.007	

***=P<0.001

3.3.5.2 Merueshi dry season - binary logistic regression results

Cook's distance influence statistics and an anomalously high leverage value and standardized residual for one case in this dataset suggested an outlier. However, on inspection of the data, it was unclear why this was so. There appeared to be no anomalous values in the original data, thus the point was clearly not a biological outlier so there was no valid reason for removing it. Nonetheless, the model was run both with and without the point to clarify its influence, but the overall model outcome was the same. Results (Table 3.6) are presented for the model including all data points.

The model had a non-significant result overall (χ^2_7=12.247, P=0.093) and had a good fit (Hosmer & Lemeshow; χ^2_8=3.510, P=0.898). However, both the Cox & Snell R^2 and Nagelkerke R^2 values were very low (0.041 and 0.072 respectively), indicating that the model is only able to account for <10% of the variation seen. Only biomass was significantly related to grazer presence.

Table 3.6 Results of the binary logistic regression model for investigating presence/absence of grazers on Merueshi in the dry season (N=294).

Independent variables	*B ± S.E*	*Wald*	*Sig.*		*Exp(B)*	*95% C.I.s for Exp(B)*
Livestock densities	.000 ± .001	.011	.916		1.000	.997 – 1.003
Distance to boma	.000 ± .000	2.693	.101		1.000	1.000 – 1.001
Biomass	.006 ± .002	7.048	.008	**	1.006	1.001 – 1.010
%CP	.107 ± .229	.219	.640		1.113	.710 – 1.745
%DOM	-.234 ± .210	1.243	.265		.791	.524 – 1.194
% green	.101 ± .180	.313	.576		1.106	.777 – 1.573
Distance to water	.000 ± .000	2.870	.090		1.000	1.000 – 1.000
Constant	9.804 ± 10.274	.911	.340		18098.348	

**=P<0.01

3.4 DISCUSSION

Ecological theory (see BurnSilver, Boone & Galvin 2003) emphasizes the logic of spatially extensive movements across a heterogeneous environment, in order to make best use of spatially separated key resources. However, in order to utilise the potential offered by a heterogeneous environment, it is essential that animals are free to move where they choose. Within an open-access environment, free of constraints by fences or other physical barriers, wild grazers should distribute themselves optimally within the landscape. Essentially, this means locating themselves in areas where they can maximise their energy gains (Bailey *et al.* 1996), within the natural constraints imposed by abiotic factors such as slope and distance to water. In reality however, there are likely to be a variety of interacting factors which play a role in determining where wild grazers are to be found.

Identification of landscape-scale determinants of large herbivore distribution patterns is an important challenge facing wildlife conservationists (Redfern *et al.* 2003). An understanding of these determinants enables the prediction of herbivore aggregations, impacts and movements, and can be very useful in designing and managing wildlife conservation areas and corridors. Logistic regression models provide an ideal way to investigate the relative importance of the different determinants.

In this study, the wild grazer guild was selected as the dependent variable for which to investigate predictors of distribution. This was because wild grazers make up the bulk of the wildlife population in the study area (74% on Mbirikani and 68% on Merueshi), and the quality and quantity of the grass resource was more easily investigated than that of browse. In addition, wild grazers are likely to be those species most in competition with livestock (mostly grazers) due to the shared forage resource, so would provide the best example of competition or displacement by human activities should there be any.

3.4.1 Comparison of landscape heterogeneity

Landscape heterogeneity, expressed as the diversity of plant communities and habitat types (Bergstrom & Skarpe 1999), is of the utmost importance when investigating the distribution of and habitat utilisation by wild herbivores (McNaughton & Georgiadis 1986; Ben-Shahar 1995). In a comparative study such as this, showing that the two areas to be compared are of ecological similarity and of equal heterogeneity is important in order to be confident that the results obtained are not biased by unexplored underlying differences in the two landscapes. I found no evidence of significant differences in heterogeneity between the two ranches, thus making it possible to relate wild grazer distribution patterns to distinct, measurable variables such as grass quality and quantity.

3.4.2 Background on the grazers

The majority of the number of grazers per square kilometre was comprised of zebra and wildebeest (60% and 34% respectively). Thomson's gazelle constituted 5% of the overall grazer density, oryx only 1%, and hartebeest less than 1%. On Mbirikani Group Ranch, zebra and wildebeest were present in fairly equal densities (45% and 49% of the total respectively), whilst on Merueshi, zebra constituted the greatest proportion of grazers (74% as compared with 34% for wildebeest). A brief description of the ecology of each species is given below.

Zebras are non-ruminants, which tend to eat large quantities of grass, including vegetation too fibrous and low in protein for ruminants to digest (Estes 1997). They are able to crop tall, tough grass as well as shorter grasses, but because their digestive system is relatively inefficient, zebras need to spend a large proportion of the day grazing, simply in order to consume an adequate amount of herbage (Estes 1997). The plains zebra is water dependant (Western 1975) and must drink frequently, although it is a misconception that zebra need to drink on a daily basis. Brooks (2005), using GPS collars to monitor zebra movements in Botswana in 2003, found that zebras foraged without water for a mean period of four days throughout the dry season. He found that zebras regularly spent over 100 hours without drinking (maximum 170 hours). Additionally, the mean maximum foraging distance from water was 17.5km, with a maximum of 34.5km (Brooks 2005).

The common (white-bearded) wildebeest is adapted to feeding on short grass (Estes 1997; Wilmshurst *et al.* 1999). Wildebeest need to drink daily, or at most every other day and are thus considered extremely water dependent. Estes (1997) reports that this dependency on water confines wildebeest to within 10-15km of water.

Coke's hartebeest graze selectively on leafy perennial grasses (Estes 1997; Ego, Mbuvi & Kibet 2003). Hartebeest will drink regularly where water is available, but can subsist on roots and tubers where no water is available (Estes 1997), and are thus considered a water independent species.

Fringe-eared oryx are capable of surviving in waterless wastelands (Estes 1997) and are entirely water-independent (Western 1975), although they do drink opportunistically (pers. obs.).

Thomson's gazelles are usually confined to short grasslands (Estes 1997). They are generally considered to be water dependent (Western 1975), and can travel up to 16km to water every other day (Maddock 1979). However, it has been observed that some individuals remain on waterless plains, so they must have the capability of water-independency (Estes 1997).

3.4.3 Regression analyses

On Mbirikani Group Ranch during the wet season, grazers were positively associated with areas of higher grass biomass and greener grass, whilst a higher percentage of digestible organic matter (%DOM) in the grass significantly decreased the odds of grazer presence. Grass greenness can be considered a proxy for grass quality, and may be a better predictor than percentage crude protein (%CP) since it was collected at the transect level and is therefore available at a much finer resolution than %CP, which was only available at the habitat level. In the wet season on Mbirikani therefore, wild grazers were selecting areas which had both high quantity and quality of grass (see below for a discussion of the negative relationship with %DOM). Grass characteristics therefore, were more important than abiotic or human related variables in explaining the observed distribution of wild grazers on Mbirikani in the wet season. That the grazers were able to distribute themselves optimally within the landscape in relation to grazing resources illustrates the openness of the Mbirikani ecosystem, and suggests freedom of movement for the wildlife.

The negative relationship between wild grazer density and the organic matter digestibility of grass is not easily explained, and merits further investigation. It is possible that selecting for a more digestible grass was less of a priority than selecting for biomass and greenness, possibly because the majority of grazers were zebras (bulk feeders) for which quantity is more important than quality.

In the other three scenarios (Mbirikani dry season and Merueshi wet and dry seasons), only grass biomass emerged as a significant predictor, being positively related to the presence of wild grazers. This might seem contradictory to much of the literature, which suggests that many species of wildlife (especially wildebeest and Thomson's gazelles) prefer areas of lower biomass (Maddock 1979; Sinclair 1985; Fryxell & Sinclair 1988; Estes 1997), as a lower biomass often indicates better food quality (Bergstrom & Skarpe 1999; WallisDeVries, Laca & Demment 1999). However, in this study, on both ranches, there was a fairly low biomass throughout the landscape, and so selecting areas of higher biomass really equates to simply choosing areas where there actually was some grass, rather than where there was virtually none. This may also help explain why grazers did not actively select areas of better quality grass; it is likely that there was so little grass available, that they must go where they could find grass to graze, irrespective of its quality. This is supported by other studies. For example, Sinclair (1974) found that in the dry season in the Serengeti, buffalo had to expand their diets to include lower quality grass components since they could not maintain their minimum nutritional intake rate by selecting only rare high quality grass. In addition, zebra need to eat considerable quantities of grass in order to fulfil their nutrient requirements (Estes 1997). Since zebras constitute 74% of the grazers on Merueshi Group Ranch, this might help to explain why it is grass biomass, rather than quality which is consistently positively associated with the presence of wild grazers. Indeed Sinclair (1985) found that in the dry season in the Serengeti, zebra preferred areas with the tallest grass.

Redfern *et al.* (2003) predicted that herbivores in African savannah ecosystems must meet their nutritional requirements within the constraints of water availability, and found this to be the case for all water-dependent or semi-water-dependent species in the Kruger National Park, South Africa. This study however, found no evidence that distance to water constrained the distributions of wild herbivores, even in dry seasons. Since 99% of the overall grazer density in this study consisted of water dependent species, this result merits attention.

There are several possible explanations as to why grazers were not found to have a significant relationship with water availability. These include the lower forage availability close to water, the influence of livestock and human activities close to water, and the influence of the actual distances involved. Irregular water availability in semi-arid savannas can affect the distribution, quality and quantity of food available for large herbivores (McNaughton & Georgiadis 1986; Bergstrom & Skarpe 1999). That forage availability is more rapidly depleted near water sources is well established (Ibrahim 1993; Johnson 1993). It may be therefore, that grazers in this study were compromising closeness to water for forage availability. Indeed, Bergstrom & Skarpe (1999) found only a very weak relationship between wildebeest density and distance to the water pans in the Kalahari, and proposed that this may be due to wildebeest being large-bodied grazers in big groups which were unable to find enough grazing close to the pans. This compromise would only be possible however, if the animals could walk to water when necessary. Estes (1997) reports that wildebeest can walk for 10-15km to water, and Thomson's gazelles 16km. The major concentrations of grazers on Mbirikani in the dry seasons were located between 20 and 26km from most available surface water, but much closer to the safari lodge water hole which may have supplemented their water requirements between long treks to the rivers (pers obs.). As reported by Brooks (2005) however, zebra can forage at much greater distances from water (up to 34.5km).

Because of the highly intensive use of all available permanent water in the study area by livestock and people, herders may have intentionally or unintentionally scared away wildlife (de Leeuw *et al.* 2001). For example, in Australia, Andrew & Lange (1986) found that kangaroos avoided areas close to water points which were used intensively by sheep, and de Leeuw *et al.* (2001) found that wildlife in northern Kenya was found much further away from water than livestock, and that wildlife assemblages were more diverse further from water. They suggested that this may be because livestock and human activities associated with water points negatively affects the distributions of wildlife (de Leeuw *et al.* 2001). Andrew (1988) suggested that animal species diversity might peak at some intermediate point along the gradient away from water. This probably applies to density of wildlife as well and is likely to be the case in this study, hence the lack of any significant relationship with wild grazer presence and distance to water.

3.4.3.1 Multiple linear regression results (from Appendix 3A)

As a second stage of analysis, multiple linear regressions were carried out on wild grazer density data, once all transects with zero observations had been removed. The detailed results are presented in Appendix 3A, but I will give a summary here to support the results presented so far. On Mbirikani in the wet season, where wild grazers were present, their densities were positively correlated with those of livestock, as well as with areas of greener grass (N=147). Once again this illustrates that wild grazers on Mbirikani were able to move freely, selecting for the best grass, and were not negatively influenced by livestock, since they congregated in the same areas, most likely because these were the best grazing areas. No predictors emerged as significant in explaining the density distribution of Mbirikani grazers in the dry season (N=81). For Merueshi, no additional information was obtained from the multiple linear regressions on actual densities. In the wet season (N=68), only biomass emerged again as significant, and no predictors were found to be significant in the dry season (N=42).

The lack of significant predictors emerging from the models, which was consistently the case in all preliminary investigation of the data, suggests that other factors may be involved. Additionally, the accuracy and applicability of the data used in the regressions could be improved, and the dependent variable could be split into individual species, rather than combining five very different species into a single variable. Suggestions for improving the model outcomes are discussed in the following section.

3.4.4 Problems and suggestions for improvement

Even in the best model (Mbirikani dry season), less than 50% of the variation in wild grazer distribution was accounted for by the single significant predictor, grass biomass (R^2=0.445). For Merueshi in the dry season, less than ten percent of the variation seen was explained by biomass, which was also the only significant predictor (R^2=0.072). This suggests that there are other factors involved which were not investigated. These could include predation or the distribution of salt licks, which are discussed below, or factors such as shade and temperature (Kennedy & Gray 1993).

Predation has been found to be important in structuring the community of herbivores in the Serengeti-Mara region of East Africa (Sinclair 1985). In this study however, the data on carnivore densities were scarce for Mbirikani and absent from Merueshi, so an indicator of predation could not be included in the model. Salt licks, which provide much needed minerals to wildlife, are recognised as an important determinant of wildlife distributions (Case 1938). For example, in Sabah, Malaysia, salt licks are believed to affect the distribution of the Sumatran rhinoceros (Lee, Stuebing & Ahmad 1993), and the distribution of elephants in Zimbabwe was found to be related to environmental sodium (Weir 1972). In fact in the US, the artificial distribution of salt has been used successfully to manage the distribution of wildlife (Case 1938). In this study, there was no information available on location of salt licks and concentrations of minerals which precluded investigation of the effect of these in explaining the observed distribution patterns of wild grazers.

For a more thorough investigation of what affects wild grazer distributions therefore, both predation and the availability of salt licks should be included. Even so, using 'wild grazers' as the dependent variable has its own inherent problems. One of the main questions being investigated was whether the grazers are able to select areas based on the best forage available. However, for ruminants such as wildebeest, critical forage *quality* thresholds are higher than for non-ruminants (e.g. zebra) which require a greater *quantity* of grass (Redfern *et al.* 2003). Therefore, combining ruminants and non-ruminants which require different grass properties complicates the analysis and may be masking important results. In addition, combining water-dependent and non-water-dependent species may complicate the accuracy of the distance to water variable, although this should be minimal since only 1% of the grazer density was comprised of non-water-dependent species. Also, combining herbivores of such different body sizes might further compromise the results. This is because small herbivores such as Thomson's gazelles have relatively higher energetic demands than larger bodied species, and are thus forced to select higher quality grasslands (Demment & Van Soest 1985; McNaughton & Georgiadis 1986). Larger species on the other hand have to expand their diets to include lower quality, more abundant plant material (McNaughton & Georgiadis 1986). For all these reasons, further investigation should be done with the distribution of individual species analysed separately. This would also help to understand the role of inter-specific competition between grazers (see Sinclair 1985).

There are several additional suggestions which may help to improve the regression analyses. A larger sample size would be beneficial, especially for the regressions in the dry season when so few wild grazers were actually present. In addition, it is possible that the data on grass quality (%CP and %DOM) were not collected at a fine enough resolution. They were collected at the habitat level, rather than the transect level, but there may be substantial variation within the habitat which could influence grazer distributions, making it possible that significant patterns would be missed. Moreover, the quality analysis was done on the basis of cattle faecal samples, which were considered to represent the highest quality of grass available in the area in which the cows were grazing on that day. However, there is not a complete dietary overlap by livestock and wild grazers, so this may not be a fair representation of the quality of grass available to wild herbivores. Nonetheless, Ego *et al.* (2003) calculated the dietary overlap between cattle and two wild grazers (wildebeest and Coke's hartebeest) to be over 70%. Lastly, it would be interesting to use production of grazing species as the dependent variable, rather than densities. This is because production, which is a function of the mean kcal equivalent of adult mass (Western 1983) better represents the forage off-take by the grazers.

Despite these suggestions for improvement, several important points have been highlighted in this study. Evidence suggests that where forage quantity is limited, grazers may have no choice but to select the areas of highest grass biomass simply in order to get sufficient forage. This suggests that forage quality may be less important than quantity, i.e. an area with a small amount of good quality grazing may be insufficient to meet the nutritional demands of grazers, which must therefore seek areas with more grass available, albeit at the expense of quality. Where maximum distances to water are less than 26km, the distribution of surface water does not seem to affect grazer distributions significantly. This may be because the elasticity of intrinsic constraints (Owen-Smith 1993) allows grazers to increase their foraging distances when the benefits provided by the distant forage outweigh the costs of travel (Brooks 2005).

3.4.5 Conclusion

Understanding the spatial dynamics of landscape use by free-ranging grazers is critical for ecosystem management (Senft *et al.* 1987; Bailey *et al.* 1996). Regression models of grazing behaviours can implicate environmental factors by which animals select preferred feeding areas (Senft *et al.* 1983). Use of these can therefore provide important information regarding the determinants involved in the observed distribution patterns of species or guilds of species.

Results from this study do not support the hypothesis that subdivision and sedentarisation of pastoralists constrains the optimal distribution patterns of wild grazers. Little evidence was found to suggest that on Merueshi grazers were negatively influenced by human factors any more so than on Mbirikani. Instead, grazers in all scenarios were found to select areas with greater grass biomass. This suggests that, in an ecosystem where grass biomass is generally low and therefore grazers are extremely resource stressed, they may have to forfeit an avoidance of human disturbances and the convenience of proximity to water, in order to find sufficient grazing.

The wet season results for Mbirikani indicate that grazers were less resource stressed in this season, thus being able to select better quality grass rather than just quantity, but also suggest that wild grazers were free of constraints on their movement. Further research into the effect of human-induced constraints versus selection of forage characteristics in a resource-limited environment would be valuable.

This chapter has investigated one potential ecological constraint of land subdivision; the effect of people on the wildlife distributions. The next chapter looks at the effect of wildlife on the local people. It takes an economic perspective to examine the degree to which the presence of wildlife imposes a cost on pastoral households.

CHAPTER 4

THE COSTS AND BENEFITS OF WILDLIFE TO MAASAI PASTORALISTS

Hypothesis: For a Maasai pastoralist in the Amboseli-Tsavo Ecosystem, the cost of living with wildlife greatly exceeds income from current wildlife revenues.

ABSTRACT

Wildlife can impose considerable costs on pastoral livestock farmers who share their land with it, and realistic quantification of these costs are required to improve management of these problems. A questionnaire survey was carried out on 177 respondents in six groups on two Maasai group ranches, to gather information on perceived losses of livestock to predation, wildlife-related diseases, drought, other diseases and other causes. Wildlife-related costs from disease and predation alone cost households on average \$585 per year, although this differed significantly between regions. Benefits amounted to a mean of only \$190 per household per year on one ranch and \$0 on the other. The deficit costs to wildlife (costs minus benefits) varied significantly between regions, with one group making, on average, a small profit from wildlife, while households in another group lost on average over \$950. None of the costs (wildlife or non-wildlife-related) differed significantly between ranches, suggesting governance is not an important factor determining livestock losses. Non-wildlife-related costs were significantly higher than wildlife-related costs from disease and predation alone, but a calculation of the scale of grazing competition by wildlife indicates that this could impose a considerable cost.

4.1 INTRODUCTION

4.1.1 Background

Wildlife and its conservation can add both opportunities and constraints to pastoral livestock farmers on whose land wildlife resides. Advantages of living with wildlife include both direct financial benefits from tourism and indirect social and aesthetic benefits (Norton-Griffiths *et al.* in press). Direct benefits, usually generated as cash for households, include wages, education bursaries, money from lodge leases and bed-night fees, income from sales of crafts to tourists and in some cases consumptive use of the wildlife resource. Indirect benefits, which tend to advantage communities as a whole, include construction of infrastructure such as roads, dams and clinics, and support for churches and schools. However, for individuals, direct benefits appear to be much more important (Norton-Griffiths 1998), and indeed indirect, community-level benefits are often not even perceived to be related to wildlife (Gadd 2005).

The costs of living with wildlife for livestock farmers include competition for forage and water, predation of livestock by carnivores (and/or costs to prevent predation), transmission of diseases to livestock by wildlife (and/or costs to prevent disease transmission) and destruction of private property by game animals (Norton-Griffiths 1996; Muthiani 2001). For agro-pastoralists there is also the significant cost of crop damage by wildlife (Deodatus 2000; Wang, Curtis & Lassoie 2006). These costs potentially make retention of wildlife on one's land, alongside livestock, very uneconomical and farmers who share their land with wildlife frequently see it as a pest and seek ways to eliminate it (Grootenhuis 2000; Marker, Mills & Macdonald 2003).

In Kenya over 70% of the wildlife is found outside protected areas, mostly on arid to semi-arid rangelands (Grunblatt *et al.* 1995b; Norton-Griffiths 1998). These same rangelands support around 25-35% of Kenya's people (Ng'ethe 1993; Ottichilo *et al.* 2000) and more than half of Kenya's livestock population (Ng'ethe 1993; Government of Kenya 1994). Clearly therefore, wildlife conservation is largely dependent on the cooperation of the local people. Whilst Maasai and other pastoralists traditionally lived alongside wildlife (Prins 1992; Prins & Grootenhuis 2000), increasing human population and consequent pressure to raise the productivity of the land, as well as higher personal expectations (Kock *et al.* 2002), means tolerance of wildlife and its associated costs is rapidly decreasing. Since hunting was banned in Kenya in 1977, drastically decreasing the opportunity for local landowners to realise any benefits from their wildlife, wildlife has declined by 60-70% (Norton-Griffiths 2007), and continues to do so, at a rate of 3-4% per annum (Kock *et al.* 2002).

Since protected areas are too small to conserve Kenya's wildlife resource effectively (Western & Gichohi 1993), the fate of the wildlife in Kenya relies on the maintenance of wildlife populations on community lands. This in turn relies heavily on the generation of sufficient revenues from wildlife to make it a valuable asset worth protecting (Prins & Grootenhuis 2000). Indeed some authors believe that communities would actively seek to preserve wildlife if they possessed a legal and valuable stake in the resource (Western

1982; Gibson & Marks 1995). Unfortunately in Kenya, the current ban on any consumptive use of wildlife denies landowners much of the potential revenue from wildlife (Norton-Griffiths 2007), but nonetheless efforts must be made to share the revenue generated from photo-tourism with the communities which support wildlife on their land.

Costs from wildlife can be considerable for pastoralists who depend on livestock for survival. Whilst wildlife revenues undoubtedly go *some* way to alleviating *some* of these costs for *some* people, in only 5% of Kenya's rangelands are any wildlife revenues generated at all (Norton-Griffiths *et al.* in press). This chapter investigates the situation on two Maasai group ranches within the wildlife dispersal area of Amboseli, Tsavo and Chyulu Hills National Parks. Specifically, this chapter aims to investigate the following hypothesis

"For a Maasai pastoralist in the Amboseli-Tsavo Ecosystem, the cost of living with wildlife greatly exceeds income from current wildlife revenues".

Costs and benefits from wildlife are quantified and discussed. The resulting information can be used both by the Maasai and by local conservation bodies to decrease costs, increase benefits and thus enhance the value of wildlife, increasing the potential for using the land for a mixed wildlife-livestock production system. The final conclusions are relevant not only to the ranches studied but to the rest of Kenya's Maasailand and indeed other parts of Africa and even the world, where pastoralists share their land with wildlife.

4.1.2 Research objectives

This chapter has the following specific research objectives.

- To quantify household livestock losses to predation, wildlife-related diseases, drought, other diseases and other causes on Mbirikani and Merueshi Group Ranches.
- To calculate wildlife revenues both paid to and received by the Maasai community on Mbirikani Group Ranch.
- To investigate the extent to which the benefits from wildlife offset the costs at both the household and group ranch level.

4.2 METHODS

A semi-structured interview was used to collect the majority of information on the costs and benefits of living with wildlife. Due to constraints on the length of the interviews, only dry-area livestock farming costs were considered, and not the costs incurred by crop farmers. Section 4.2.1 details what information was included and what was omitted, and Section 4.2.2 gives details of the diseases included in the study.

4.2.1 Choice of information to gather

Predation: Predation includes not only carnivore predation but also incidents of livestock deaths from buffalo, elephants and baboons, as well as the costs of preventative measures taken to reduce predation. Cost of injury alone was not included.

Wildlife-related disease: The livestock diseases considered as 'wildlife-related' were east coast fever (ECF), malignant catarrhal fever (MCF) and trypanosomiasis for cattle, and kurru-nkonyek for sheep and goats ('shoats'). The latter has no English translation but is an eye disease of shoats, poorly understood but believed to be spread by gazelles. Included in the cost of wildlife-related diseases are the costs of drugs or chemicals used in prevention or cure. Two potentially important costs which are not included are the forced sale of livestock in poor condition due to disease (Cleaveland *et al.* 2000; Bedelian, Nkedianye & Herrero 2007), and costs of lowered output due to poor health.

Drought: This includes deaths from water deprivation, lack of grazing and other deaths considered by the Maasai to be as a direct result of drought. Not included is the important cost of reduced output caused by drought-induced weakness, owing to the difficulty of obtaining accurate information on this.

Other disease: Any disease that was not one of the four mentioned above falls into this category. The major diseases of cattle were contagious bovine pleuro pneumonia (CBPP), anthrax and lumpy-skin disease. For shoats, the major diseases reported were contagious caprine pleuro pneumonia (CCPP), enterotoxaemia (red-intestine disease), anthrax and a capripox virus related to the lumpy skin disease of cattle.

Other cause: Losses included in this category incorporate deaths from swallowing plastic bags, accidental injury leading to death, vehicle related deaths, birth complications, getting stuck in the mud and vulture attacks.

The survey did not include information about donkeys. There were very few donkeys on the ranches in comparison to cattle and shoats (pers. obs.), they rarely died of disease and there was no economically significant disease spread to donkeys from wildlife. Thus the only important consideration was donkey predation. Accurate information on this was obtained from the Predator Compensation Program which had been operational on Mbirikani since April 2003 (S. Maclennan, unpublished). Also unaccounted for are the indirect costs of wildlife, such as exclusion from national parks and deliberate avoidance of wildlife dense areas in an attempt to avoid predation or disease transmission.

No attempt was made to quantify costs incurred from human death and injury caused by wildlife. This is because of the difficulty in obtaining accurate information on a clearly sensitive topic. Such injuries are frequently not reported in any official capacity, making it impossible to obtain accurate information from Kenya Wildlife Service or even local clinics and hospitals, and Maasai are usually unwilling to discuss this in interviews. Whilst some studies have attempted to record incidents of human death or injury (e.g. Treves & Naughton-Treves 1999; Sitati *et al.* 2003), this is usually done for a single species or over a relatively small area. Moreover, quantifying *costs* from reported wildlife-inflicted injury can be extremely subjective and is rarely attempted, and to attempt to quantify the cost of human life is far beyond the scope of this project.

4.2.2 Description of diseases included as 'wildlife-related' diseases

This section describes the wildlife-related diseases included in this study, and briefly explains for each disease the role of wildlife in its epidemiology. This information is later used to determine the percentage contribution by wildlife to the cost of each disease.

4.2.2.1 East Coast Fever (ECF)

ECF is a tick-transmitted protozoal disease of cattle, considered to be one of the most important diseases affecting cattle in East Africa (Uilenberg 1995). The protozoan parasite, *Theileria parva* is transmitted by the tick *Rhipicephalus appendiculatus* (Norval, Perry & Young 1992). Whilst the original host for *Theileria parva* is believed to have been the buffalo, ECF can be largely maintained by cattle (Grootenhuis 2000). However, while the control of ECF in cattle can be largely achieved by tick control with acaricides (D'Haese, Penne & Elyn 1999), no such control can applied to wildlife. Thus wildlife maintains a tick burden that is outside the control of acaricides applied by farmers and is consequently an important factor in the maintenance and spread of ECF. However, this tick burden from wildlife is comparatively small. On a ranch in Kenya with an equal biomass of livestock and wildlife, despite being sprayed twice per month with acaricide, the cattle still maintained approximately 75% of the total tick population on the farm, with wildlife maintaining only 25% of the ticks (Jonyo *et al.* 1986). Grootenhuis (2000) estimates the cost contribution by wildlife to be 10-50% for ECF.

4.2.2.2 Malignant Catarrhal Fever (MCF)

MCF is a fatal disease of cattle caused by the alcelaphine herpes-virus-1 (Grootenhuis 2000). Wildebeest are the wildlife reservoir (Cleaveland *et al.* 2000; Bedelian *et al.* 2007). Although other wildlife species have been shown to harbour antibodies, only wildebeest calves under 4 months old are able to transmit the disease to cattle (Rweyemamu *et al.* 1974; Mushi, Rurangirwa & Karstad 1981). It is not maintained in the livestock population, cannot spread from cow to cow, and occurrence is limited to areas where the distribution of cattle and wildebeest coincide (Grootenhuis 2000; Bedelian *et al.* 2007).

4.2.2.3 Trypanosomiasis

Trypanosomiasis is caused by infection with protozoan organisms of the genus *Trypanosoma*, transmitted by tsetse flies (Grootenhuis 2000). It is a disease of huge economic importance to livestock farmers (Itty 1993), to the extent that endemic trypanosomiasis and efficient livestock production are not compatible (Grootenhuis 1999; Hursey 2001). Wildlife however can thrive under heavy trypanosomiasis challenge and several wild Bovidae are maintenance hosts of trypanosomes (Grootenhuis 1999). The contribution to costs of trypanosomiasis by wildlife is dependent on the relative wild to domestic animal densities (Grootenhuis 2000).

4.2.2.4 Kurru nkonyek

The veterinary knowledge of this disease is scarce, but vets and interviewees both believe it to be spread to sheep and goats from gazelles. It affects the eyes of the animals and can lead to blindness (G. Thurasha, *pers. comm.*).

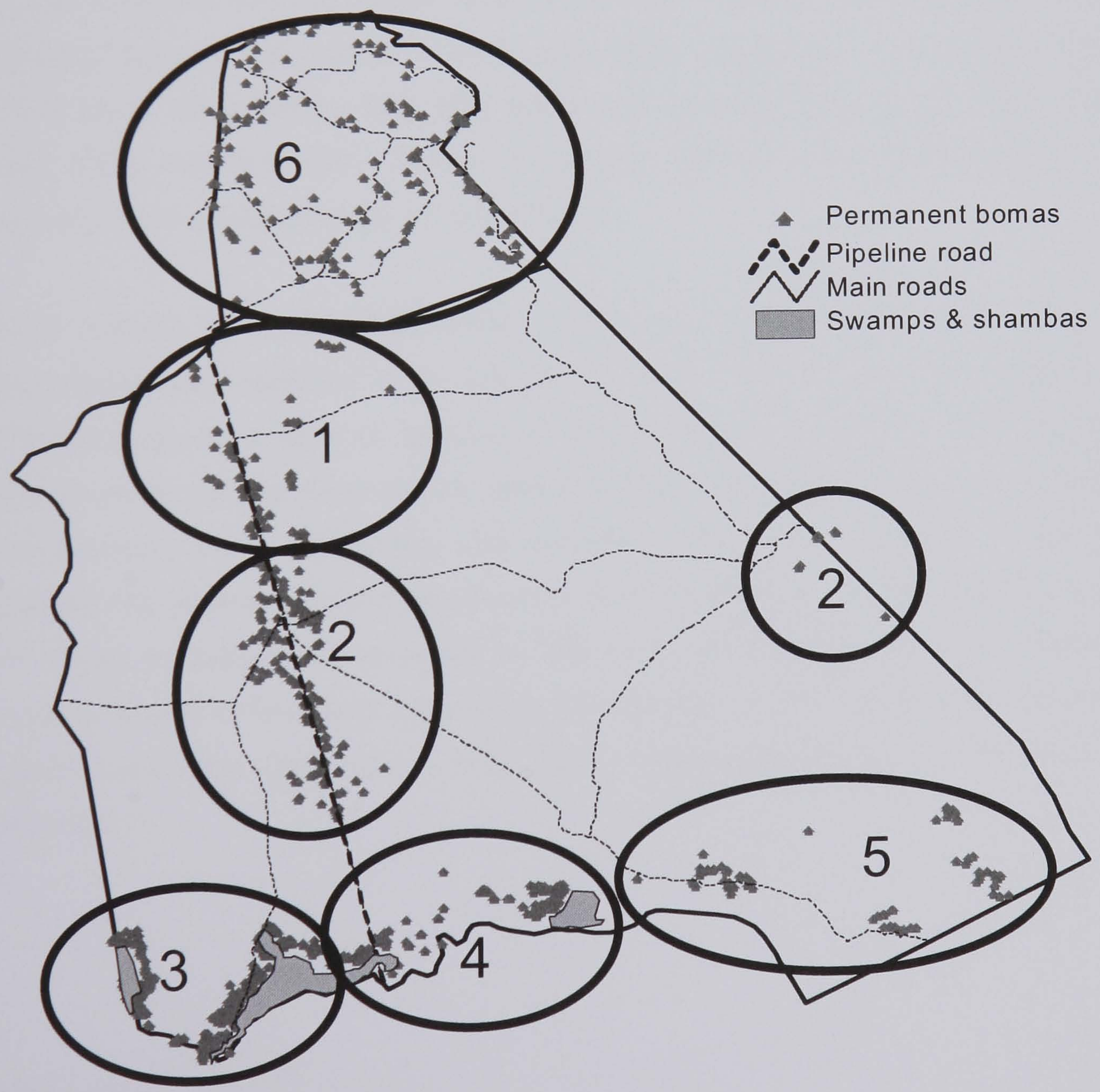
4.2.3 Questionnaire design and interview methodology

Guidelines for the design of the questionnaire and the structure of the survey were taken from Robson (2002) and White *et al.* (2005). The questionnaire used can be found in Appendix 4A. The population to be sampled comprised all male household heads on Mbirikani and Merueshi Group Ranches (although on occasion corroboration of answers was sought with other community members). A household was defined as a married man, all his wives and unmarried children, and anyone else who depended on him for food and shelter (often an elderly relative or the offspring of a deceased relative). This is the basic unit of production and decision making in Maasai society (Bekure *et al.* 1991). Stratified random sampling was used on this population, whereby the population was divided into a number of groups, with each group sharing a particular characteristic (Robson 2002). In this case the stratification was based on attitudes to wildlife and level of interaction with wildlife which were broadly known before the survey started, such that the sample population represented a continuum of severity of conflict with wildlife (Sutton, Larson & Jarvis 2004).

Five groups were chosen on Mbirikani (groups 1-5). The whole of Merueshi Group Ranch was taken as one group because it was very small, completely subdivided and all the members seemed to have similar attitudes towards and experiences with wildlife. The six groups had different numbers of households, varying between 87 and 321 but since responses within each group showed little variation, a set number of 30 questionnaires were carried out in each group (although three were later discarded). Group 2 on Mbirikani was split between two different locations as the inhabitants had permanent homesteads in both areas. The groups are illustrated in Figure 4.1 and background information given in Table 4.1 (Results Section). Each group was scored as very poor, poor, average, good or very good for employment or agricultural opportunities based on expert knowledge and their distance to towns and rivers respectively. The mean distances of each group from water and towns were calculated using the nearest-features extension in ArcView GIS (v3.2), and scored as far (>10km), close (<5km) or very close (<1km).

Respondents were chosen randomly from a complete list of all possible options in each group. The names of every household head were collected prior to starting the survey and a random number generator (Excel 2003) was used to choose the interviewees. The questionnaire took one to two hours to complete and there was a 100% response rate. Each respondent was scored for co-operation, knowledge and honesty in order to be able to check for bias due to less co-operative or knowledgeable respondents and also to potentially discard interviews which were felt to be very dishonest. Two Maasai enumerators helped to carry out the questionnaires. Both had previous experience in conducting interviews and both were well known and respected members of the community.

Figure 4.1 Map of Mbirikani and Merueshi Group Ranches showing how the permanent bomas were grouped for the questionnaire survey.



Before the survey was started, comprehensive pre-testing was carried out and the interview went through several changes before being finalised. These pre-tests (N=12) were also used to correct interviewer bias and both enumerators were comprehensively trained in how to conduct the interview to avoid any bias. The questionnaire required both quantitative and qualitative responses but all apparently open-ended responses were coded at the time into a series of pre-chosen answers (Neuman 2003). This list of potential answers was built from the pre-test interviews, and an 'other' category was always present.

4.2.4 Treatment of questionnaire results

The questionnaire asked for information about bulls, steers, cows and calves separately as well as rams, castrated rams, ewes and lambs, and billy goats, castrated billy goats, adult female goats and kids. The mean market price for each category was used to calculate the cost of losses and the overall value of the herd. Values used are as follows (in Kenyan shillings): bulls 16,000; steers 14,000; cows 9000; calves 4000; rams and castrated rams 2,300; ewes 1000; lambs 500, billy goats and castrated billy goats 2,700, adult female goats 1000 and kids 500. Values in Kenyan shillings were converted to US$ at the February 2007 exchange rate of US$1=Ksh70.

Visual outliers were identified during initial examination of the data. Each of these respondents was revisited in an attempt to clarify their answers and double check the information given, a form of 'ground-truthing' (White *et al.* 2005). If the same answers were given a second time round, and a satisfactory explanation given for the original inconsistency, the questionnaire was included in the overall analysis. If the respondent changed his answers, further verification was sought from sons, neighbours, herd-boys and wives to judge the accuracy of the information given. Where it was clear the respondent was deliberately misleading the interviewer and the data were inaccurate, the questionnaire was discarded. Only three questionnaires out of 180 were ultimately discarded.

4.2.5 Additional methods

4.2.5.1 Calculating the costs of competition for grazing with wildlife

The grazing costs imposed on Mbirikani and Merueshi Group Ranches by wild herbivores were calculated using the methodology of EcoSystems Ltd (1978). Metabiomass densities of wild herbivores (the mean metabolic weight on an area) were used to calculate a value for the grazing lost to wildlife. The occupancy of the ranches by wild and domestic herbivores was estimated by a year of monthly population counts using strip and point transects (Chapter 2) and these population estimates were used to calculate the cost of forage utilisation by wild herbivores. Metabiomass was calculated by raising species' mean weights (taken from Western 1973) to the power of 0.75 (EcoSystems Ltd. 1978): the use of this particular scaling exponent is discussed below. Multiplying a species' mean metabiomass (a common unit of analysis) by the density of that species present in the designated area gave total metabiomass. By knowing the mean metabiomass of wild herbivores on a ranch during a particular period, it was possible to estimate the number of cattle that could have been there in place of the wildlife. The value of those cattle could be considered fair compensation for the support of that wildlife (EcoSystems Ltd. 1978).

In Chapter 2, wildlife production was calculated using a scaling function of body mass to the power of 0.67, as opposed to 0.75 used here. Whilst 0.75 is a commonly used scaling exponent (Demment & Van Soest 1985; Savage *et al.* 2004), currently there is a considerable debate in the literature about exactly what the scaling function should be (Glazier 2005; White & Seymour 2005), and there seem to be valid arguments for use of either 0.66-0.68 or 0.75 (Beuchat *et al.* 1997). Clauss *et al.* (2007) have recently published an excellent review on this issue. They conclude that dry matter intake scales to body mass (BM) at $BM^{0.77}$ for ruminant foregut fermenters and $BM^{0.64}$ for caecum fermenters. An inclusion of colon fermenters and non-ruminant foregut fermenters gives an average of $BM^{0.76}$ (Clauss *et al.* 2007). However, White and Seymour (2005) give an allometric exponent of 0.68 if large herbivores are excluded. Glazier (2005) specifically states that, whilst the 0.74 power law remains valid, other metabolic scaling relationships occur which are equally as valid. For the purposes of this study, where the bulk of the herbivore population under consideration consists of medium sized herbivores, with an almost equal number of ruminants and non-ruminants, the use of different scaling factors should have little effect on the outcome (D. Western, *pers. comm.*), and I have therefore followed the methods of the respective studies without standardising the scaling factor.

4.2.5.2 Informal interviews with key informers

Richard Bonham (owner of Ol Donyo Wuas safari lodge) and Fred Njagi (Ol Donyo Wuas Trust manager) were interviewed informally to get information relating to wildlife revenues and trust activities, as were the lodge managers and administrators. Certain educated and informed Maasai were also used to corroborate and explain some information from the questionnaires.

4.2.5.3 Use of Predator Compensation Fund data

The Predator Compensation Fund (PCF) was initiated in April 2003. Through a system of predator scouts, verification officers and the PCF team, any member of Mbirikani Group Ranch could report, and be compensated for, any livestock killed on the ranch by a predator within the previous 24 hours (S. Maclennan *pers. comm.*). Since the majority of people want compensation for their killed livestock (pers. obs.), the PCF reports give a good picture of predation fitting the PCF criteria. These data (S. Maclennan, unpublished) were used to compare with the questionnaire results and additionally to get an estimate for the number of donkeys predated.

4.3 RESULTS

Table 4.1 gives summary statistics and background information for the six survey groups. Group 5 was the most marginal group with the highest levels of conflict with wildlife, while groups 2 and 3 were closest to major infrastructure and had lower interactions with wildlife.

Table 4.1 Background information for the six survey groups (shoat = sheep and goats combined). Groups 1-5=Mbirikani, group 6=Merueshi.

Data	***1***	***2***	***3***	***4***	***5***	***6***	***Mean (±SE)***
Total number of households per group	87	206	321	146	173	160	182 ±32.0
Number of interviews used	30	30	30	28	30	29	30 ±0.3
(% of households interviewed)	(34%)	(15%)	(9%)	(19%)	(17%)	(18%)	(16%)
Mean household size	14	11	10	10	13	12	12 ±0.6
Mean cattle holding per household	93	55	34	48	50	46	54 ±7.5
Mean shoat holding per household	106	82	42	86	101	75	82 ±6.2
Mean livestock holding per household	199	137	76	134	151	120	136 ±12.9
Mean years of education of household head	2	4	3	1	1	2	2 ±0.3
Mean attitude to wildlife (1=negative, 5=positive)	3.5	3.2	3.2	3.0	2.5	1.8	2.9 ±0.3
Density of habitat	medium	medium	open	open	dense	medium	
Interaction with wildlife	average	average	low	high	v. high	average	
Employment options	good	v. good	v. good	poor	v. poor	v. poor	
Agricultural opportunities	none	none	v. good	good	poor	poor	
Distance from water	close	v. close	v. close	close	far	far	
Distance from town	close	v. close	v. close	close	far	far	

4.3.1 Wildlife-related diseases and predation; adjusting the costs

This section describes how reported costs of disease and predation were adjusted to give a fairer estimate of the cost contribution by wildlife. Whilst ECF, MCF, trypanosomiasis and kurru-nkonyek can all be considered wildlife-related, for both ECF and trypanosomiasis, which are maintained and spread by both cattle and wildlife, only a certain proportion of the cost can be attributed to wildlife. It is important to adjust the cost estimate for this contribution to avoid an overestimation of the role of wildlife in the cost of that disease. The outputs of the multiple linear regression models investigating which factors affected the prevalence of these diseases (Appendix 4B) could have been used to determine what percentage of the costs of each disease were attributable to wildlife. However, the data was not collected in sufficient depth to allow such detailed use of the results, which showed a different proportion of disease costs from wildlife and livestock than was suspected to be the case. A combination of my knowledge of the situation, expert advice from local veterinarians and reports in the literature were therefore used to estimate the contribution of wildlife to the costs of the diseases.

For predation, the reported losses were scaled down to represent only those losses which occurred when the livestock were being looked after properly, to avoid blaming predators for what was effectively a loss due to careless livestock husbandry.

East Coast Fever (ECF)

Grootenhuis (2000) estimated the cost contribution by wildlife to the prevalence of ECF to be 10-50%. On Mbirikani Group Ranch, the biomass of cattle was twice as great as that of wildlife (and even greater on Merueshi). In addition, there were no buffalo recorded on Merueshi Group Ranch and very few on Mbirikani. It is therefore likely that the vast majority of ECF was maintained and transmitted by the livestock population itself. Therefore, for this study area, the contribution by wildlife to the costs of ECF is taken as 10%.

Malignant Catarrhal Fever (MCF)

Since MCF is spread to cattle entirely from wildebeest, and is neither maintained nor spread by the cattle population itself, wildlife is taken to contribute 100% of the cost of this disease.

Trypanosomiasis

As mentioned, the contribution to costs of trypanosomiasis by wildlife is dependent on the relative densities of wildlife and livestock in the area. Since livestock densities exceeded wildlife densities in the study area by 3-8 times (see Chapter 2 for details), it is likely that trypanosomiasis, as for ECF, was also maintained and transmitted largely within the livestock population itself. A regression analysis investigating the factors associated with disease prevalence (Appendix 4B) supports this supposition. It shows that wildlife did not play a significant role in affecting the prevalence of trypanosomiasis, while livestock did. For these reasons, the contribution of wildlife to the cost of trypanosomiasis was taken to be 10%.

Kurru nkonyek

Kurru nkonyek was not an economically significant disease. Due to the lack of understanding of this disease, 100% of the costs incurred were attributed to wildlife, since this was perceived to be the case by the Maasai themselves.

Predation

The Mbirikani Predator Compensation Project which ran concurrently with this study recorded all livestock kills reported, allocating penalties for poor husbandry; either losing livestock in the bush or having livestock taken from a poorly constructed boma (S. Maclennan, *pers. comm.*). No penalties were allocated if the depredation event occurred when the livestock were being properly herded, or if the animal was taken out of a well-constructed boma. Therefore the ratio of livestock deaths penalised to not-penalised gave an estimate for the proportion of livestock deaths that could be fairly blamed on wildlife. The calculated values show that only 20% of cattle killed could reasonably be blamed on wildlife; the remaining 80% of kills occurred as a result of poor husbandry (S. Maclennan, unpublished data). Therefore only 20% of the costs of reported predation on cattle were attributed to wildlife. For shoats however, 75% of those killed were being looked after properly at the time (S. Maclennan, unpublished data) and thus 75% of the cost of all reported shoat deaths were attributed to wildlife.

Table 4.2 gives the mean costs of total reported losses to wildlife predation and individual diseases (A) followed by the adjusted costs (B). Both reported and adjusted costs still represent people's perceptions. The cost of the total reported losses are useful for a comparison with other studies where values are not adjusted. However, it is the adjusted costs that are used in the remainder of this chapter since they represent a more realistic cost to wildlife and are therefore more useful from a management perspective.

As illustrated in Table 4.2, by adjusting for husbandry (i.e. not including predation events that were due to human negligence), overall mean costs to predation were reduced by 57% ($199 to $85). By reducing the cost contribution of wildlife to ECF and trypanosomiasis by 90% each, the overall mean cost of wildlife-related disease was reduced by 75% ($1738 to $434).

Table 4.2 Mean (±SE) reported and adjusted costs of predation and wildlife-related diseases. Values in parentheses represent the percentage of total losses included in the adjusted cost estimate. Total N = 177.

	COSTS - US$/hh/yr Group (sample size)	***1 (N=30)***	***2 (N=30)***	***3 (N=30)***	***4 (N=28)***	***5 (N=30)***	***6 (N=29)***	***Overall Mean ± SE***
A	Reported cattle predation	146.7 ± 3.8	77.6 ±22.4	61.4 ±48.8	64.3 ±23.8	198.1 ±50.1	146.3 ±63.7	116.1 ±18.4
	Reported shoat predation	82.6 ± 6.7	111.6 ±51.8	34.5 ±18.3	78.3 ±20.9	110.6 ±26.3	78.7 ±24.8	82.8 ±11.9
	Total reported cost to predation	*229.3 ±54.1*	*189.2 ±59.9*	*95.9 ±65.2*	*142.6 ±37.0*	*308.7 ±66.1*	*225.0 ±73.1*	*198.9 ±24.9*
	Reported ECF	520.0 ±131.62	646.7 ±163.4	575.2 ±287.7	1013.8 ±258.1	2355.7 ±516.6	784.2 ±377.2	983.4 ±136.5
	Reported MCF	325.2 ±109.6	165.7 ±79.1	47.1 ±25.0	294.9 ±200.5	519.1 ±216.2	295.6 ±213.9	274.3 ±64.2
	Reported trypanosomiasis	78.1 ±46.8	0.0 ±0.0	384.8 ±297.4	825.5 ±173.5	1225.2 ±268.5	295.6 ±153.8	465.1 ±83.7
	Reported kurru nkonyek	19.7 ±9.3	5.9 ±5.9	1.7 ±1.7	21.8 ±8.3	24.2 ±9.8	19.0 ±12.2	15.3 ±3.5
	Total reported cost to wildlife disease	*943.0 ±192.6*	*818.2 ±203.9*	*1008.9 ±534.2*	*2156.0 ±560.5*	*4124.2 ±842.4*	*1394.4 ±710.2*	*1738.0 ±241.7*
	Total reported cost	**1172.3 ±224.7**	**1007.4 ±207.1**	**1104.8 ±589.8**	**2298.6 ±585.1**	**4432.9 ±873.2**	**1619.3 ±769.5**	**1936.9 ±256.2**
B	Adjusted cattle predation (20%)	29.3 ±8.7	15.5 ±4.5	12.3 ±9.8	12.9 ±4.8	39.6 ±10.0	29.3 ±12.7	23.2 ±3.7
	Adjusted shoat predation (75%)	62.0 ±12.5	83.7 ±38.9	25.9 ±13.8	58.7 ±15.7	83.0 ±19.7	59.0 ±18.6	62.1 ±8.9
	Total adjusted cost to predation	*91.3 ±18.5*	*99.2 ±39.9*	*38.1 ±22.7*	*71.6 ±18.0*	*122.6 ±25.7*	*88.3 ±24.7*	*85.3 ±10.7*
	Adjusted ECF (10%)	52.0 ±13.2	64.7 ±16.3	57.5 ±28.5	101.4 ±25.8	235.6 ±51.7	78.4 ±37.7	98.3 ±13.7
	Adjusted MCF (100%)	325.2 ±109.6	165.7 ±79.1	47.1 ±25.0	294.9 ±200.5	519.1 ±216.2	295.6 ±213.9	274.3 ±64.2
	Adjusted trypanosomiasis (10%)	7.8 ±4.7	0.0 ±0.0	38.5 ±29.7	82.6 ±17.4	122.5 ±26.8	29.6 ±15.4	46.5 ±8.3
	Adjusted kurru nkonyek (100%)	19.7 ±9.3	5.9 ±5.9	1.7 ±1.7	21.8 ±8.3	24.2 ±9.8	19.0 ±12.2	15.3 ±3.5
	Total adjusted cost to wildlife disease	*404.7 ±115.3*	*236.2 ±236.2*	*144.9 ±72.7*	*500.6 ±231.5*	*901.3 ±265.2*	*422.5 ±260.8*	*434.4 ±78.1*
	Total adjusted cost	**496.0 ±127.5**	**335.4 ±90.0**	**183.0 ±90.4**	**572.2 ±244.4**	**1023.9 ±279.1**	**510.8 ±275.8**	**519.7 ±83.1**

4.3.2 Including costs of defensive activities in the total cost estimate

Results so far have dealt only with the costs of cattle and shoat deaths from various causes. There are however additional expenses which need to be considered when calculating the total costs from wildlife. These include the defensive activities (costs of disease prevention and treatment and prevention of predation) and costs of donkey predation (taken from PCF). These extra costs are given in Table 4.3, along with the costs of livestock deaths from wildlife to provide the final mean cost from wildlife.

Table 4.3 Breakdown of mean costs from losses to, and prevention and treatment of, predation and wildlife-related disease. The value of $20 for prevention of predation is the mean expenditure required to build a strong, predator-proof boma and the value of $2 for donkey deaths is the total annual loss on the ranch (from PCF data) divided by the total number of households.

	TOTAL COSTS (US$/hh/yr)	*1*	*2*	*3*	*4*	*5*	*6*	*Mean (±SE)*
Deaths	Cattle deaths - disease	385.1	230.4	143.1	478.8	877.1	403.6	419.1 ± 76.9
	Shoat deaths - disease	19.7	5.9	1.7	21.8	24.2	19.0	15.3 ± 3.5
	Cattle deaths - predation	29.3	15.5	12.3	12.9	39.6	29.3	23.2 ± 3.7
	Shoat deaths - predation	62.0	83.7	25.9	58.7	83.0	59.0	62.1 ± 8.9
	Donkey deaths - predation	2.0	2.0	2.0	2.0	2.0	2.0	2.0
Defensive activities	Prevention & treatment - cattle disease	54.3	36.9	25.2	40.8	48.6	39.2	40.8 ± 3.1
	Prevention & treatment - shoat disease	1.7	0.9	0.4	2.8	2.7	3.8	2.0 ± 0.4
	Prevention of predation	20.0	20.0	20.0	20.0	20.0	20.0	20.0
	Total (± standard errors)	**574.0 ± 134.2**	**395.2 ± 92.0**	**230.6 ± 95.4**	**637.8 ± 248.9**	**1097.1 ± 288.3**	**575.7 ±282.2**	**584.5 ± 85.5**

From Table 4.3 it can be seen that households were losing on average $585 per year from wildlife. The major costs were from cattle deaths from disease, and households spent a mean of $41 per year on prevention and treatment of these diseases. Overall, wildlife-related disease cost significantly more than predation (log T_{176}=-6.805, P<0.001).

4.3.3 Benefits

This section briefly covers the wildlife revenues generated on the two ranches. A more detailed case study of the revenues generated by Mbirikani Group Ranch is found in Chapter 5. Mbirikani supported one safari lodge and was the beneficiary of an active conservation trust associated with the lodge. Both these generated revenues for the ranch. Wildlife benefits received by Mbirikani households, as reported in the interviews, averaged US$190 per household per year (Table 4.4). In contrast, Merueshi Group Ranch had no tourism industry or trust and these households received no monetary benefits from wildlife.

Table 4.4 Summary of wildlife benefits received by Mbirikani Group Ranch members as reported in the questionnaire survey. Values are in US$ per household per year, ± standard errors. N=148.

Benefit Type	*1*	*2*	*3*	*4*	*5*	*Mean*
Employment	47.4 ± 47.4	122.9 ± 71.6	195.4 ± 97.9	33.7 ± 25.8	82.9 ± 59.3	97.3 ± 29.4
Education bursaries	21.0 ± 9.5	235.7 ± 217.2	12.4 ± 4.5	0.0 ± 0.0	2.9 ± 1.6	55.1 ± 44.1
Predator compensation	33.8	33.8	33.8	33.8	33.8	33.8
Other	4.8 ± 4.8	0.0 ± 0.0	10.0 ± 5.6	0.0 ± 0.0	3.3 ± 2.5	3.7 ± 1.6
TOTAL	107.0 ± 52.3	392.4 ± 275.9	251.6 ± 98.6	67.5 ± 25.8	122.9 ± 59.1	189.9 ± 61.7

4.3.4 Total cost-benefit analysis

Table 4.5 shows the costs, benefits and the deficit costs to wildlife by group. This highlights the major disparity between groups in costs and benefits to wildlife. For example group 5 households lost almost $1000 per year to wildlife disease and predation, whilst group 3 households actually profited from wildlife. On average, households in group 2 lost virtually nothing to wildlife, having a deficit cost of only about $3 per household per year. However, these averages mask huge differences between individual households, illustrated by the large standard errors.

Table 4.5 Wildlife-related costs and benefits, and the cost-deficit by group. Figures are means ± standard errors in US$ per household per year. χ^2 and P are results of Kruskal-Wallis tests for differences between groups. Total N = 177.

	1	*2*	*3*	*4*	*5*	*6*	χ^2 *(df = 5)*	*P*
Costs	574.0 ± 134.2	395.2 ± 92.0	230.6 ± 95.4	637.8 ± 248.9	1097.1 ± 288.3	575.7 ± 282.2	28.716	<0.001
Benefits	107.0 ± 52.3	392.4 ± 275.9	251.6 ± 98.6	67.5 ± 25.8	122.9 ± 59.1	0.0 ± 0.0	102.753	<0.001
Deficit	467. 0 ± 150.3	2.8 ± 258.4	-21.0 ± 139.8	570.2 ± 252.8	974.3 ± 295.7	575.7 ± 282.2	27.234	<0.001

Post-hoc testing showed that for costs, group 3 was significantly different from groups 1, 2, 4 and 5. With group 3 removed, there was no longer a significant difference between groups (KW: χ^2_4=8.708, P=0.069). Unsurprisingly, for benefits, group 6 was significantly different from all other groups, but if group 6 is removed, there was no longer a significant difference between the groups (KW: χ^2_4=8.278, P=0.082). For deficit cost, group 3 was significantly different from groups 1, 4, 5 and 6, but not 2. These considerable differences illustrate the importance of location in determining overall costs to wildlife.

4.3.5 Comparison between Merueshi and Mbirikani Group Ranches

In previous analyses data have been presented in six groups. Groups 1-5 were all on Mbirikani Group Ranch and group 6 represented Merueshi. In this section, data for groups 1-5 are combined to represent Mbirikani for the purpose of an inter-ranch comparison. This illustrates the effect of land-use policy (subdivided versus communal) on wildlife-related costs and benefits. Table 4.6 gives a summary of the costs of livestock losses to all causes by ranch, with results of the statistical comparison of differences between the two. The high mean values relative to the median values indicate that most people suffered fairly low costs whilst a few suffered much greater losses. The latter were often the households with large livestock holdings.

For none of the causes of livestock deaths did costs differ significantly between ranches. When the same tests were carried out for cattle and shoats separately, the only significant difference between ranches was for costs of 'other causes' for shoats (W=2069.5, Z=-2.44, P=0.015), with Merueshi having significantly higher costs than Mbirikani. Although non-significant, values indicate that Merueshi households suffered higher costs from drought than Mbirikani households, but lower costs from both wildlife-related and other diseases. Costs from predation were similar on both ranches.

Table 4.6 Comparison of costs of livestock deaths (cattle and shoats combined) between Mbirikani and Merueshi Group Ranches. Values are means ± standard error, with medians below, in US$ per household per year. W and Z statistics and P values are from Mann-Whitney U-Tests.

Cause	*Mbirikani* *N=148*	*Merueshi* *N=29*	*W*	*Z*	*P*
Predation	84.74 ± 11.89 32.14	88.26 ± 24.70 19.28	2473.0	-0.441	0.659
Wildlife related disease	436.70 ± 78.67 102.14	422.51 ± 260.82 38.57	2143.5	-1.747	0.081
Drought	652.99 ± 116.70 109.29	1215.81 ± 595.11 185.71	13060.5	-0.464	0.643
Other disease	882.20 ± 138.34 222.86	687.39 ± 219.25 332.86	13075.0	-0.386	0.700
Other cause	65.57 ± 9.49 0.00	102.27 ± 36.18 0.00	2510.0	-0.313	0.754

4.3.5.1 Inter-ranch cost-benefit analysis

Costs from wildlife were significantly higher than benefits on both Mbirikani and Merueshi Group Ranches (Z=-7.770, P<0.001, n=148 and Z=-4.703, P<0.001, n=29 respectively). Table 4.7 shows there were no significant differences in overall costs to wildlife between ranches. Moreover, the overall deficit cost to wildlife did not differ significantly between ranches, despite the revenues generated by Mbirikani.

Table 4.7 Costs and benefits from wildlife, plus deficit, by ranch, in US$ per household per year. Figures are means ± standard errors. Mann-Whitney U-Test results for significance of differences between ranches are indicated as W, Z and P values. Since all Merueshi household received zero benefits, it was not possible to statistically test the inter-ranch difference in benefits.

	Mbirikani	*Merueshi*	*W*	*Z*	*P*
Cost from wildlife	586.25 ± 86.53	575.69 ± 282.19	2368.00	-0.844	0.399
Benefit from wildlife	189.90 ± 61.66	0.00 ± 0.00	435.00	____	____
Deficit	396.35 ± 105.03	575.69 ± 282.19	12995.00	-0.701	0.483

4.3.6 Comparison of wildlife and non-wildlife-related costs

Previous results have demonstrated the disparity between the costs and benefits from wildlife. However, to put wildlife-related costs into perspective, it is useful to compare these with non-wildlife-related costs such as drought, diseases which are not linked to wildlife and other causes of livestock death such as eating plastic bags, birthing problems and accidental deaths. Appendix 4C gives details of the costs of all these causes to cattle and shoats independently and by group. A summary is presented in Table 4.8.

Table 4.8 Summary table of costs from wildlife-related and non-wildlife-related causes (deaths only). Values are in US$ per household per year ± standard errors, with results of paired t-tests on logged variables. N=177

	Wildlife-related costs	*Non-wildlife-related costs*	*T (df=176)*	*P*
Cattle	442.3 ± 78.6	1202.3 ± 192.1	-3.558	<0.001
Shoats	77.4 ± 10.3	464.8 ± 58.4	-12.613	<0.001
Livestock	519.7 ± 83.1	1667.1 ± 237.2	-9.693	<0.001

Non-wildlife-related costs were significantly higher than wildlife-related costs for cattle, shoats and livestock overall. For cattle, total wildlife-related costs were 3 times lower than non-wildlife-related costs. The most important non-wildlife-related cost for most groups was drought, followed by disease (mainly CBPP and anthrax). For shoats, wildlife-related costs were six times lower than non-wildlife-related costs. This is mainly because there were no economically significant wildlife-related diseases for shoats, while non-wildlife-related diseases such as lumpy skin disease, CCPP and enterotoxaemia constituted a considerable cost.

4.3.7 Competition for grazing

Grazing competition by wildlife is often considered to constitute a considerable cost to pastoral households (Mizutani *et al.* 2005) and is an important consideration when investigating overall costs from wildlife. Competition for grazing can be approximated by calculating the proportion of the grass utilised by wildlife, as compared with livestock. This calculation is illustrated for Mbirikani Group Ranch in Appendix 4D.

On average, wildlife on Mbirikani Group Ranch consumed 31.5% of the grazing resource available; the remaining 68.5% was utilised by livestock. This varied little seasonally, being 31.1% in the wet season and 33.1% in the dry season. Assuming wildlife to be direct cattle equivalents, one could say that wildlife costs the group ranch 31.5% of the value of the total livestock herd. In 2005, the total livestock herd (using figures reported in the questionnaire) was worth $12,441,555. Thirty-one and half percent of this is $3,919,089, so theoretically, the cost of grazing competition from wildlife could be almost $4 million. This amounts to a mean loss of $4287 per household per year to grazing competition. Considering the restricted dietary overlap between wildlife and livestock however (Croze *et al.* 1978), the actual cost is likely to be considerably lower.

On Merueshi Group Ranch, wildlife consumed only 15.8% of the grazing resource. This result is consistent with the finding that Merueshi supported approximately half the density of wildlife found on Mbirikani (Chapter 2).

4.4 DISCUSSION

The cost data presented in this study are based almost exclusively on what people have said in interviews or informal conversations, and it is therefore accepted that what is presented is *perception* rather than *fact.* Furthermore, there was an assumption that the Maasai respondents could accurately recall livestock-related issues, especially deaths, for the previous year. The analysis of perceived loss data can be justified by the fact that individuals tend to act on their perceptions, rather than factual information, making this the most important determinant of attitude and behaviour (Mishra 1997; Moberly *et al.* 2003). Additionally, since livestock play such a central role in the Maasai household, it is likely the respondents would have remembered events fairly accurately (see Bedelian *et al.* 2007). However, if not entirely accurate, it is assumed that the information given represents an overestimate of losses rather than an underestimate (Catley 2003; Hazzah 2007).

Nonetheless, figures generated in this study are consistent with other studies. For example, this study found that total livestock losses to predation (before adjusting for husbandry) averaged 3.5% of the herd. A study by Patterson *et al.* (2004) on ranches adjacent to Tsavo National Park in Kenya reported an annual loss of 2.4% of the livestock herd to predation, and in Zimbabwe in 1995, 5% of livestock holdings were reported killed by predators (Butler 2000). Additionally, total losses to predation reported in this study were not significantly different from the losses recorded by the Mbirikani Predator Compensation Fund for the same time period, which were calculated as 2.3% of the herd

(S. Maclennan, *pers. comm.*). Prevalence of disease recorded in this study is also consistent with other studies. For example, Plowright *et al.* (1975) reported that under optimal conditions for the MCF virus, outbreaks may affect only 7% of the exposed population, and in a study in the Ngorongoro District of Tanzania, 5.6-6.2% of cattle in high risk villages died from MCF (Cleaveland *et al.* 2000). In this study, in an ecosystem where only a small area presents a high risk of MCF, MCF was reported to have affected on average 3.4% of the cattle herd. These consistencies promote confidence in the results obtained.

4.4.1 Adjusting the costs of wildlife-related disease and predation reports

It is important to be accurate and realistic about wildlife-related costs, because in general, disease risks from wildlife have been overestimated (Grootenhuis 2000), as have the costs of predation, and in the past this misconception has led to massive eradication of wildlife to control disease (Wooff 1968 in Grootenhuis 2000) and determined persecution of predators (Marker *et al.* 2003). Data on the economic consequences of disease transmission between livestock and wildlife are almost non-existent, and the costs of disease to the livestock owner in Africa have previously been poorly documented (Grootenhuis 2000). Detailed research in this field, with costs and benefits quantified at both the household and ranch level, could be of enormous practical value.

There are many diseases that can, at least in part, be attributed to wildlife and Grootenhuis (2000) provides an excellent review. However, trying to get details of a large number of individual diseases from Maasai farmers is likely to be difficult and may reduce the accuracy of information given (Billiouw *et al.* 2002; Sutton *et al.* 2004). In addition, the three cattle diseases chosen (ECF, MCF and trypanosomiasis), with the possible exception of foot and mouth, are the only diseases maintained by wildlife that are of major economic importance (Itty 1993; Grootenhuis 1999; Cleaveland *et al.* 2000). Nonetheless, both ECF and trypanosomiasis are spread and maintained by both cattle and wildlife, and it was therefore necessary to estimate what proportion of the disease was likely to be attributable to wildlife and scale-down the reported costs accordingly.

Adjusting the reported costs from predation to include only those cases where livestock were killed despite good husbandry is an unusual procedure. It is of course a fact that all events of livestock depredation are due to wildlife, and it is important not to ignore the total costs from predation. However, with improved livestock husbandry (such as efficient herding and building strong, predator-proof bomas), incidents of depredation could be significantly reduced (Ogada *et al.* 2003). Indeed Kruuk (1980 in Prins 2000) concluded that "*the most important factor causing exposure of livestock to predation is human negligence*". Disregarding the losses from predation that were due to human negligence allows a cost estimate which represents the underlying level of predation in a situation where people's herding practices are efficient and their bomas well-built. This may be considered a 'true cost' to predation and could be a useful value for determining potential compensation. It is important to note however, that improved husbandry may incur a cost in itself, through defensive activities.

Results from Table 4.2 (part A) show that, before adjustment, ECF was the disease that cost households the most, followed by MCF then trypanosomiasis. This is consistent with the findings by Cleaveland *et al.* (2000), who found that for Maasai pastoralists in the Ngorongoro Distict, Tanzania, ECF was the disease of most concern, with MCF among the five most important cattle diseases. However when adjusted for the contribution by wildlife (i.e. a 90% reduction for both ECF and trypanosomiasis), MCF emerged as the most important wildlife-related disease.

Before adjusting for husbandry, predation of cattle cost households more than predation of shoats (Table 4.2 (part A)). However, only 20% of the cattle that were lost were killed despite proper husbandry. For shoats however, 75% of kills occurred despite good husbandry, thus when costs are adjusted for husbandry, predation of shoats emerged as more important than that of cattle (Table 4.2 (part B)). The difference can be attributed to the fact that shoats are much smaller and herd sizes were often bigger than for cattle, both of which make them easier targets for predators (especially hyaenas), even when being properly herded (Mizutani *et al.* 2005). For cattle, the majority of predation events occurred when they were lost and therefore left outside the boma at night (S. Maclennan, unpublished), and so these losses were attributed to human negligence rather than predators *per se*.

4.4.2 Cost-benefit analysis

4.4.2.1 Costs

There was a significant difference in costs from wildlife between groups, with group 3 having significantly lower costs than all other groups except group 6. This is probably because group 3 had considerably smaller livestock holdings than all other groups (Table 4.1), and so losses to wildlife were bound to be lower. Moreover, this group was located in a region with very little wildlife, and because they were close to water, they had less need to travel extensively with their herds, risking contact with wildebeest calves (carriers of MCF) and predators. Additionally, there were no tsetse flies in that area, and the short grasses were less optimal for ticks. The same reasons help explain why group 2 also had a relatively low cost from wildlife.

Group 5 suffered significantly higher costs than group 3, and with a wildlife-related cost of over $1000 per household per year, this result merits attention. Group 5 had the highest costs from both wildlife-related disease and predation independently. It is likely that two factors combine to explain these results. Firstly, households in group 5 had the largest cattle herds (pers. obs.), although during the questionnaire survey these were grossly under-reported (see Table 4.1). A study by Hazzah (2007) found that households in this particular group under-reported their herd sizes by an average of 100 cattle. However, this group was an exception and reported herd sizes in the other groups were much more accurate (from personal verification). If it is understood that group 5 households had much larger cattle herds than all the other groups, the higher costs incurred by these households are more easily explained. However, it is also the case that households in group 5 (and to a lesser extent group 4), did have the greatest problems with wildlife and disease. Predator concentrations in those areas were the highest on the ranch (S. Maclennan, *pers. comm.*), and the density of the surrounding vegetation made the area a hot-spot for disease vectors such as tsetse flies and ticks. In addition, group 5 was closest spatially to the main wildebeest calving areas, where MCF was prevalent. During discussion sessions with the community about the findings from this study, all groups agreed that households living in group 5 had the most problems from both disease and predation.

The major differences in wildlife-related costs within Mbirikani Group Ranch suggest that environmental factors were very important in determining the extent of wildlife-related livestock losses. Results presented in Appendix 4B support this.

4.4.2.2 Benefits

Wildlife-related benefits differed significantly between groups because group 6 received no financial revenues from wildlife. Within the five groups on Mbirikani however, there was no significant difference in benefits received. Nonetheless, benefits did differ considerably between groups on Mbirikani, ranging from a mean of $68 to $392 per household per year. The high variation within these samples (illustrated by the high standard errors in Table 4.5) may account for the lack of significant differences, but the variation itself is an important point. Indeed these mean values hide the fact that only 24% of households actually received any financial benefits from wildlife. Additionally, the high mean wildlife-related benefits reported for group 2 was largely due to one individual. It is noteworthy that the groups with the highest wildlife-related costs were also the groups with some of the lowest benefits (Table 4.5).

4.4.2.3 Cost-benefit analysis

Mean deficit costs to wildlife (net costs minus benefits) clearly illustrate the extreme disparity in wildlife-related costs and benefits between groups. On average, households in group 3 actually profited from wildlife, while group 5 households lost almost $1000 dollars per year to wildlife. The costs and benefits for group 2 almost completely balanced each other, while group 1, 4 and 6 lost around $500 on average per household. It would be expected therefore that attitudes towards wildlife would differ considerably between these groups, which has important implications for conservation. Chapter 5 explores this in detail.

4.4.3 Comparison between Merueshi and Mbirikani Group Ranches

A comparison between Mbirikani and Merueshi Group Ranches is relevant in order to investigate the impact of land subdivision on wildlife-related costs and benefits. Whilst livestock densities did not differ significantly between the ranches, wildlife densities did, with Merueshi supporting only half the density of wild macro-herbivores found on Mbirikani (see Chapter 2 for details).

Table 4.6 shows there were no significant differences in any cause of livestock death between Mbirikani and Merueshi Group Ranches. Considering the significantly lower wildlife densities found on Merueshi Group Ranch, it may seem surprising that neither predation nor wildlife-related diseases were significantly lower than on Mbirikani. However, Merueshi Group Ranch was not a closed system and livestock was frequently grazed on neighbouring ranches (including Mbirikani) and in the Chyulu Hills National Park, where it would have come into contact with higher wildlife densities, including buffalo, wildebeest and predators. This suggests that decreasing wildlife will not necessarily lead to a decrease in wildlife-related costs unless the system is self-contained and/or livestock husbandry is improved. Since significant differences were found in wildlife-related costs between groups within Mbirikani, the lack of significance between the two ranches overall suggests that it was factors other than land use policy which were most important in determining costs from wildlife.

Whilst both ranches suffered considerable costs from wildlife, only Mbirikani received any benefits. However, there was still no significant difference in the deficit cost to wildlife between the ranches (Table 4.7). Despite a mean annual income from wildlife of almost $190 per household on Mbirikani Group Ranch, wildlife costs were still considerably higher, with mean wildlife revenues accounting for only 32% of the costs from wildlife-related disease and predation alone. Even so, unfortunately, the situation on Mbirikani Group Ranch today is still the exception rather than the rule: over 95% of Kenya's rangelands generate no wildlife revenues of any sort (Norton-Griffiths & Butt 2006). The situation on Merueshi Group Ranch is therefore more representative of the majority of Kenya's rangelands.

4.4.4 Comparison of wildlife and non-wildlife related costs

On average, livestock deaths from wildlife-related-disease and predation cost households $520 per year (Table 4.8). However, costs from non-wildlife related causes were found to be significantly higher, costing households over three times as much as the wildlife-related costs. Costs of diseases attributed to wildlife were found to be less than costs of other diseases, especially CBPP and anthrax, and drought was the cause of considerable losses to some households. This is consistent with findings by Mizutani *et al.* (2005) working on Mbirikani Group Ranch in 2002-3. In addition, although pastoral losses to predation are often given the greatest attention by western society, this study, alongside others, clearly illustrates that these costs are usually minor in comparison with disease and drought (Mizutani 1995; Mizutani *et al.* 2005).

4.4.5 Competition for grazing

This chapter has focussed on the costs from wildlife-related disease and predation, because they are easier to quantify than other costs. Nonetheless, problems from these two causes may be virtually inconsequential in comparison with resource competition, and in a study done on Mbirikani by Muthiani & Wandera (2000), competition for forage was ranked second in importance as a harmful effect of wildlife after the spread of disease. I calculated that 31.5% of the grazing resource on Mbirikani was consumed by wildlife in 2005 (see Appendix 4D for details), which is consistent with other studies. For example, de Leeuw (1991) found that wild herbivores add roughly 25-30% to the livestock biomass in the study area, while Norton-Griffiths (1996) found wildlife consumed grazing representing 30% of the livestock density. If wildlife biomass was considered directly replaceable by livestock biomass, this represents a maximum of almost $4 million in lost opportunities for livestock, or $4287 per household per year.

However, according to some, there is little or no evidence that livestock numbers are in fact reduced by high densities of wildlife (Prins 2000). The interactions between wildlife and domestic herbivores are complex and a simple competitive interaction is rare (Prins 2000). For example, diet separation, where animals use different parts of the plants, may minimize competition (Croze *et al.* 1978). Where wildlife does compete with livestock for food and water, Deodatus (2000) and Prins (2000) describe the costs of these competitive interactions as 'negligible'. This means that even if wildlife were to be removed, livestock could not expand on a one to one ratio to fill its niche. Due to this complexity, it is unusual to attempt to quantify the costs of competition for grazing (but see Heath 2000), although this does not decrease its importance.

It is certainly an important factor to consider when comparing costs between areas. For example, Table 4.7 shows that there was no significant difference in wildlife costs between Mbirikani and Merueshi when considering just predation and wildlife-related disease. However, since Mbirikani lost a far higher proportion of its grazing resource to wildlife than Merueshi (31.5% versus 15.8%), it is possible that wildlife actually did cost Mbirikani households significantly more.

4.4.6 Conclusion

The cost data presented in this study were based on perception of loss translated into a monetary equivalent for the sake of a cost-benefit analysis. However, for the Maasai, livestock are worth far more than just their monetary equivalent. They play a major cultural role and are an important indicator of social status (Spear & Waller 1993). The loss of a cow can also have an emotional cost, and the loss of a donkey can remove a household's only means of transporting water or firewood. Nonetheless, all figures presented represent only the direct financial cost.

The figures presented are a snapshot of the situation for pastoralists in Kenya today, and from the cost-benefit analysis it is easy to see why many consider wildlife a pest. In the current economic environment, many households would be considerably better off without wildlife on their land. Indeed, it is this economic environment which is impeding the expansion of the wildlife sector through major policy distortions (Norton-Griffiths *et al.* in press). These include laws which deny landowners highly profitable sources of wildlife rents from consumptive utilisation (i.e. sport hunting, cropping and bird shooting), and ownership distortions whereby owners and operators of tourism facilities divert most wildlife rents away from the landowners (Earnshaw & Emerton 2000; Norton-Griffiths *et al.* in press). It may even be that lifting the ban on consumptive use of wildlife would be one of the greatest advancements for wildlife conservation efforts, once again turning wildlife into a valuable asset to be protected. For example, Norton-Griffiths *et al.* (in press) estimate that the consumptive wildlife trade in Kenya would be worth close to $500 million if it were legal today. This illustrates the extent to which the current ban is depriving landowners of wildlife revenues and thus incentives to conserve the wildlife resource.

Although costs from non-wildlife-related causes were significantly higher than wildlife-related costs, wildlife nonetheless clearly imposes a considerable burden on many households. Intra-ranch location was found to have a significant effect on wildlife-related costs, whilst ranch did not, suggesting that environmental factors were more important than governance and land use policy. An attempt to elucidate these factors with regard to disease is presented in Appendix 4B, but further work needs to be done.

4.4.7 Summary

In general, the hypothesis that wildlife costs exceed wildlife benefits is supported by the results of this study. However, regional differences were so great that in one area, wildlife benefits did actually exceed costs. On average however, on a ranch with no wildlife-utilising infrastructure, living with wildlife imposed a net cost of approximately $575 per household per year, and clearly makes wildlife a considerable burden to already marginalised livestock farmers. Even where households earned on average $190 per year from wildlife, their net cost from wildlife was still almost $400. The current ban on all consumptive use of wildlife makes it virtually impossible for landowners to realise sufficient benefits from their wildlife to give them any incentive to conserve it. Revenues from tourism and charitable trusts alone are currently far from sufficient to offset the costs incurred from wildlife. With the increasing human pressure on the land, problems from wildlife are likely to increase, and if the western world wants to secure wildlife for aesthetic reasons, they are going to have to pay considerably more than they currently do.

This chapter has quantified the costs and benefits of wildlife to Maasai pastoralists. The next chapter investigates the effect of these benefits on Maasai attitudes to wildlife and conservation.

CHAPTER 5
THE EFFECT OF WILDLIFE REVENUES ON ATTITUDES AND BEHAVIOUR OF MAASAI PASTORALISTS IN THE AMBOSELI-TSAVO ECOSYSTEM

Hypothesis: The presence of wildlife revenues positively influences pastoralists' attitudes to wildlife but are currently insufficient to create behavioural change

ABSTRACT

Within the past two decades there has been a proliferation of attempts to establish community support for wildlife and conservation through the sharing of revenues and empowerment of local communities to manage their wildlife. This chapter presents data from two neighbouring Maasai group ranches in the wildlife dispersal area of Amboseli and Tsavo National Parks. One ranch generates considerable wildlife revenues from a tourist operation and community trust whilst the other receives no direct benefits from wildlife. Despite the overall attitude to wildlife on the former ranch being significantly more positive than the latter, there remains a spatial distinction, with attitudes varying significantly between regions depending on both costs from wildlife and the perception of the distribution of wildlife revenues. Ordinal logistic regression analyses show that it is not the amount of revenue received or even the scale of costs from wildlife which determines people's attitudes, but simply the presence or absence of wildlife benefits. The importance of addressing the inequitable distribution of benefits is emphasized.

5.1 INTRODUCTION

5.1.1 Background

The general perception in recent decades is that the inclusion of local communities in wildlife management is indispensable for successful conservation (Gibson & Marks 1995). If wildlife can generate revenues for local people, it could create positive incentives for its conservation (Child 2000; Wunder 2000). Conversely, conflict between wildlife and people can prevent or erode local support for conservation (Gadd 2005). Wildlife-based benefits are intended to offset the costs and encourage tolerance (Gadd 2005), and wildlife-based employment and development projects aspire to provide local community members with a sense of proprietorship over wildlife (Gibson & Marks 1995). However, frequently the link between benefits and wildlife is not understood, making the attempts to encourage conservation ineffectual (Child 2000; Archabald & Naughton-Treves 2001; Gadd 2005). For example, local people may be positive about tourism but remain negative about wildlife conservation (Walpole & Goodwin 2001).

There is no doubt that tourism can bring benefits to wildlife-rich areas (Western & Wright 1994; Adams & Hulme 2001). However, this does *not* automatically ensure the support of local people for conservation, as wildlife-related costs usually remain significantly higher than the benefits (Boyd *et al.* 1999; Adams & Infield 2003) and benefits are often available only to an elite minority (Thompson & Homewood 2002). This latter point is emphasised frequently in economic-based studies of Maasailand and indeed throughout Africa (Gillingham & Lee 1999; Archabald & Naughton-Treves 2001; Ogutu 2002). It is a major stumbling block to the conservationist's goal of increasing wildlife benefits to promote tolerance. In addition, the vast majority of revenues still go to operators and owners of the safari industry, rather than the communities (Gibson & Marks 1995; Norton-Griffiths *et al.* in press).

Whilst there is now an acceptance of the fact that rural communities need to participate in, benefit from and support the sustainable management of their wildlife resource (Gillingham & Lee 1999), it is still being debated how such integrated approaches might best achieve the desired results (Barrett & Arcese 1995; Bajracharya, Furley & Newton 2006). Indeed some conservationists doubt that revenue sharing can significantly improve conservation outcomes and report no positive correlation between revenue sharing and positive attitudes of the local communities towards wildlife (Parry & Campbell 1992; Gibson & Marks 1995; Hackel 1999). However, others have found that even modest revenue sharing can improve attitudes and tolerance (Lewis, Kaweche & Mwenya 1990; Archabald & Naughton-Treves 2001). Due to these complexities, surveys of rural people's conservation attitudes are an important tool during the design, implementation and evaluation stages of community based conservation schemes (Hartup 1994; Gillingham & Lee 1999).

There are a number of factors which may affect a household's perception of the costs and benefits from wildlife, including direct economic benefits, the major economic activity undertaken by the household, the local land tenure, the wealth of the household and various cultural factors (Arjunan *et al.* 2006). These are investigated in this study for two neighbouring Maasai group ranches in southern Kenya, Mbirikani and Merueshi, where the pastoralists and their livestock live alongside a diverse and relatively abundant wildlife population. Both study areas are described in detail in Chapter 1, and Section 5.1.2 gives details of the wildlife benefits available on both ranches.

The questions being examined in this chapter are whether or not the wildlife benefits available to Mbirikani Group Ranch members create a positive attitude to wildlife, and whether they are sufficient to create behavioural change in a pro-conservation direction. Do the lack of benefits on Merueshi Group Ranch result in more negative attitudes to wildlife and how does their behaviour differ from Mbirikani members? The following hypothesis is investigated:

"The presence of wildlife revenues positively influences pastoralists' attitudes to wildlife but are currently insufficient to create behavioural change".

5.1.2 Case studies

5.1.2.1 Case study 1: Mbirikani Group Ranch.

A small, luxury safari lodge (Ol Donyo Wuas) has been operating on Mbirikani since 1986 and the ranch has been the beneficiary of an active conservation trust since 1991 (Ol Donyo Wuas Trust). Members therefore had the possibility of employment at the safari lodge, or though the Trust which employed game scouts, radio operators and forestry staff. Education bursaries for secondary school and college students were provided and schools supported with resources and through payment of teachers' salaries. Conservation fees and land rents were paid by the lodge to the Group Ranch committee. In addition there was a Predator Compensation Fund (PCF) active on Mbirikani Group Ranch and a Kenya Wildlife Service (KWS) revenue sharing program from Amboseli National Park.

5.1.2.2 Case study 2: Merueshi Group Ranch

Merueshi Group Ranch had no tourist facilities or conservation trust. It received no revenue from KWS and none of the members interviewed had any kind of employment in the wildlife-sector.

5.1.3 Research objectives

This chapter has the following specific objectives:

- To quantify wildlife benefits generated on Mbirikani at both household and ranch level
- To describe people's attitudes towards wildlife and the reasons given for these attitudes
- To investigate which factors affect attitudes to wildlife
- To describe attitudes towards conservation and indicators of behavioural change

5.2 METHODS

5.2.1 Data collection

Key informants and company records provided information on wildlife revenues generated on Mbirikani Group Ranch. However, the majority of data presented in this chapter were collected using a semi-structured questionnaire survey of 177 households on Mbirikani (N=148) and Merueshi (N=29) Group Ranches. Details of the sampling stratification as well as the design and implementation of the questionnaire are given in Chapter 4, Section 4.2.3, and a copy of the questionnaire can be found in Appendix 4A. The section of the questionnaire which dealt with attitudes towards land subdivision, wildlife and perception of problems from wildlife consisted of open ended questions, with responses coded at the time into a series of pre-chosen answers and an 'other' category. Everyone who responded to the questionnaire was scored by the interviewer immediately after the interview according to the overall impression given of their attitudes to wildlife. The attitude scores ranged from one (very negative) to five (very positive), and were used as the dependant variable in the ordinal logistic regression analyses. These attitude scores were based on a combination of the respondents reported like or dislike of herbivores and carnivores, their reported desire to kill or conserve various species of wildlife, and their reported willingness to engage in pro-conservation activities, as well as extra qualitative information.

This approach is justified as it represents the best use of all available information in making the judgement of attitudes. A great deal of qualitative information was gathered during the questionnaires, outside the official remit of the questions, which provided valuable insights into the respondent's attitude to wildlife. Using a more rigorous scientific approach based only on the ranks of responses to the structured questions would be underutilising the value of the extra information obtained when doing a face to face interview. Although this is a subjective method, it was never difficult or ambiguous to assign attitude scores to respondents and I am confident there is no bias in the approach.

5.2.2 Treatment and analysis of questionnaire results

Frequency distribution data were cross tabulated into contingency tables and subjected to chi-square analysis. Where necessary, responses were combined in order to get adequate sample sizes (Weladji, Moe & Vedeld 2003). Data were analysed using the Statistical Package for Social Scientists (SPSS) v12.0, and Minitab v13. Kruskal-Wallis tests were used to investigate differences between groups because the data could not be transformed to meet assumptions of normality. For the ordinal logistic regressions, response information, goodness of fit test statistics and measures of association were all checked to ensure the model had a good fit.

5.2.3 Feedback and community discussion sessions

In late March 2007, four feedback workshops were held in different areas of Mbirikani Group Ranch, and one on Merueshi Group Ranch, to present and discuss the results of the questionnaire survey. These informal discussion sessions were used to verify the accuracy of the results obtained in the interviews and to obtain a local perspective on the interpretation of some of the results.

5.2.4 Delineations and limitations

This chapter covers the attitudes of the Maasai communities on Mbirikani and Merueshi Group Ranches towards wildlife and its conservation. The questionnaire was targeted at male household heads only, and thus all attitudes and opinions given belong to this group. Women's attitudes are not represented here, nor those of children or young men who are not yet head of the household. In addition, the focus group for the study were Maasai pastoralists and small-scale agro-pastoralists, so attitudes are not representative of the agricultural sector, nor of those living and working exclusively outside the pastoral sector. Since pastoralists tend to be more positive towards, and tolerant of, wildlife than their crop-farming neighbours (Gadd 2005), the results presented here are likely to represent the more positive end of the scale.

5.3 RESULTS

This section begins with a case study of the wildlife revenues generated on Mbirikani Group Ranch, as a background to people's perceptions of benefits and their attitudes towards wildlife and conservation.

5.3.1 Wildlife benefits generated on Mbirikani Group Ranch

According to information provided by Ol Donyo Wuas Lodge owner Richard Bonham, Ol Donyo Wuas Trust Manager Fred Njagi, Kenya Wildlife Service, African Wildlife Foundation and Mbirikani Predator Compensation Project, wildlife revenues generated for Mbirikani Group Ranch amounted to approximately US$230,450 in 2005. This is itemised in Table 5.1.

Dividing this total by the area of the ranch (321,100 acres) gives a figure of $0.7 per acre per year generated by wildlife. However, only about one third of the ranch area is used for wildlife viewing, so the revenue for land actually used by tourism is around $2.1 per acre per year.

The data in Table 5.1 also indicate that the extent of revenue sharing by Ol Donyo Wuas Lodge was high in comparison to most of Kenya. The total revenue earned by Ol Donyo Wuas Lodge in 2005 was $411,090 (ODW accountant *pers. comm.*). The total returned to the community (see Table 5.1) was $84,755 ($34,340 in conservation fees + $41,260 in wages + $9,155 in land rents), which is approximately 21% of revenues generated. In general in Kenya, landowners find it difficult to capture more than 5-10% of the wildlife rents (Norton-Griffiths & Butt 2006), less than half of what Mbirikani members were receiving.

Much of the money generated by the lodge or the trust (wages, education bursaries and compensation) was paid directly into the hands of the individual. All conservation fees and rents, however, were paid to the group ranch committee for the intended purposes of ranch administration and investment in the community. Misappropriation of funds and poor governance however, meant the vast majority of this money never reached the community. There was therefore a discrepancy between the amount of revenues generated on the ranch, and the amount actually received by the households, as illustrated below.

Table 5.1 All direct wildlife revenues generated on Mbirikani Group Ranch (MGR) in 2005. Source: Richard Bonham, owner Ol Donyo Wuas Lodge, Fred Njagi, manager of Ol Donyo Wuas Trust and Kenya Wildlife Service.

Source	***Amount (US$)***	***Comments***
Kenya Wildlife Service		
KWS bursaries	11,644	Education bursaries
African Wildlife Foundation		
AWF bursaries	6,000	School and college bursaries for MGR members from AWF
AWF wages	1,622	For employment of MGR members
Ol Donyo Wuas Lodge		
ODW wages (full time staff)	37,808	For the 25 out of 41 employed staff who are MGR members
ODW wages (casual staff)	3,452	All casual labour for 2005 (MGR members only)
ODW conservation fees	34,340	Bed night fees for 2005 (1717 bed nights)
Rents	9,155	Lease payments for lodge and Bonham and Hill households
Boma visits	5,000	Includes visiting fee ($10 per person) and craft purchases
Nyumbani wages & food	8,548	Wages of MGR members who work at Bonham household
Bird shooting	1,582	Fees paid to the MGR for bird shooting
Ride Kenya Safaris		
Rents (Ride Kenya)	3,521	Ride Kenya Horse Safari's lease for 1 year
Staff wages	1,233	3 MGR members employed initially, more to follow
Ol Donyo Wuas Trust:		
Education scholarships	23,400	School, college and university fees for 37 children
Game scouts wages	32,970	20 game scouts, 8 predator scouts and 1 verification officer
Teachers salaries	4,780	6 teachers from 4 different schools
School support	2,012	Support for 5 schools
Forestry staff	3,452	For the Trust's reforestation program
Radio operator	2,301	As a co-ordinator for the game scouts
Compensation Funds		
Predator Compensation	29,489	Paid to households to compensate for predated livestock.
Elephant Compensation	317	Paid by Amboseli Elephant Research Project
Research		
Research wages	4,954	Wages for research assistants and camp staff
Research fees	1,370	Research fees (for 2 full time researchers)
Casual employment	1,500	Employment of GR members on a casual basis for research
TOTAL REVENUE	**230,450**	

Table 5.2 shows the mean wildlife revenues reported (in questionnaires) to be received by households in each of the five groups on Mbirikani (See Chapter 4, Table 4.1 for a description of the groups). This indicates that, on average, households received approximately $190 per year from wildlife. There were 933 households on Mbirikani Group Ranch, so this amounts to a total of approximately $177,300 for the ranch ($190 x 933). Table 5.1 however, indicated that total wildlife revenues generated by Mbirikani amounted to $230,450. The deficit ($53,273) can be mainly attributed to the money paid directly to the group ranch committee in rents and conservation fees (a total of $48,386) plus the support given to schools ($2,012). This suggests that very little of the money which passed through the committee ever got to individual community members. If the money paid to the committee was instead distributed evenly among households, mean household revenues from wildlife would increase from $190 to $250 per year.

Table 5.2 Breakdown of annual wildlife revenues received according to questionnaire respondents in different groups on Mbirikani Group Ranch. χ^2 and P results are from Kruskal-Wallis tests investigating differences between groups. There is no result for predator compensation, as the data were originally averages of the whole divided evenly between the groups.

Wildlife revenues (US$/household/year)	*1*	*2*	*3*	*4*	*5*	*Mean ± SE*	χ^2	*P*
Job in tourism	47.4	28.6	121.1	0.0	82.9	56.8 ± 23.89	2.371	0.668
Education bursaries	21.0	235.7	12.4	0.0	2.9	55.1 ± 44.12	11.997	0.017 *
Predator compensation	33.8	33.8	33.8	33.8	33.8	33.8		
Other wildlife-related job	0.0	94.3	28.6	0.0	0.0	24.9 ± 14.9	5.339	0.254
Job as a game scout	0.0	0.0	45.7	33.7	0.0	15.6 ± 10.5	5.719	0.221
Craft sales to tourism	4.8	0.0	7.1	0.0	0.0	2.4 ± 1.4	5.271	0.261
Cash benefits	0.0	0.0	2.9	0.0	3.3	1.3 ± 0.8	5.249	0.263
TOTAL (± SE)	107.0 ±52.3	392.4 ±275.9	251.6 ±98.6	67.5 ±25.8	122.9 ±59.1	189.9 ± 61.7	8.278	0.082

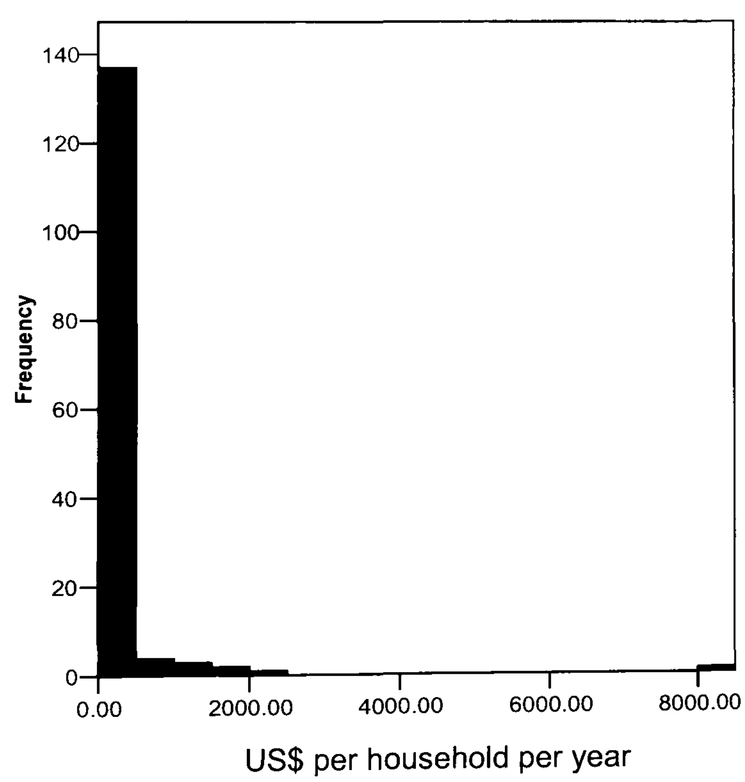
* = P<0.05

Table 5.2 shows that there was a significant difference between groups in revenue from education bursaries (P=0.017). For example, for an individual household in Group 2 (around the main village), a substantial amount of money could be received through education bursaries (although the high mean ($236) was heavily influenced by one respondent whose son was being sponsored through college in Arusha by African Wildlife Foundation, and education bursaries were on average considerably lower). With Group 2 removed, there remained a significant difference between groups in the magnitude of bursaries received (χ^2_3=10.723, P=0.013), although post-hoc testing showed no significant difference between any pairs of groups.

For households in most groups, a job in tourism was the highest money earner, and income from the Predator Compensation Fund (PCF) the next most important source of revenue, *based on averaged figures.* Group 2 households earned on average the most from wildlife, followed by group 3, the other group which surrounded a major town. Group 4 households earned the least from wildlife as this was a marginal area with few connections to the lodge or Trust.

However, the values in Table 5.2 are means, and mask the huge inequality in earnings per household. Medians for all benefit types except compensation were zero. This is highlighted in the histogram in Figure 5.1.

Figure 5.1 Frequency histogram of wildlife revenues earned per household on Mbirikani Group Ranch (N=148).



Overall, only 24% of Mbirikani households actually received anything from wildlife (excluding compensation), with the range of annual revenues being from \$48 to \$8276 per household (mean=\$693, median=\$148). The remaining 76% of households received no monetary benefit from the wildlife resource (excluding compensation). Apart from revenue from the PCF which was available to everyone on Mbirikani, education bursaries were the most widely distributed form of benefit, with 14% of Mbirikani households receiving some kind of support. Tourist jobs were held by only 5% of the entire community. Game scout jobs and cash handouts each supported 2% of the community. Additionally, indirect benefits for Mbirikani included provision of infrastructure (dams, roads and schools), a clinic, use of wild game meat and use of wildlife products such as skin, horns and tails.

Merueshi Group Ranch received no direct benefits from wildlife. They had no tourist operation, no conservation trust and received no revenues from KWS or AWF. However, 17% of respondents admitted to using wildlife products in the form of consumption of game meat.

5.3.2 Perceptions of problems associated with wildlife

An understanding of the level and type of conflict people experienced with wildlife is important when trying to understand their attitudes. To investigate these issues, people on both Mbirikani and Merueshi were asked what problems they had had with wildlife during the previous two years. Predation was perceived to be the greatest problem overall, with 92% of Mbirikani households and 76% of Merueshi households mentioning it. Resource competition was also perceived to be a major problem, especially for Merueshi households for whom it was the most important problem: 86% of households reported competition for grazing, and 66% reported problems with competition for water, whilst on Mbirikani, only 31% of households mentioned competition for grazing as a problem and only 3% reported competition for water. The issue of disease transmission from wildlife to livestock was mentioned by 55% of households on both ranches. Crop damage was reported as a problem by 52% of Mbirikani households, but only 10% of Merueshi households.

5.3.3 Attitudes to herbivores and carnivores

Overall attitudes to wildlife were significantly more positive on Mbirikani than on Merueshi (mean scores = 3.06 and 1.83 respectively; χ^2_4=25.259, P<0.001). Within Mbirikani, there was also a significant difference between attitudes by region (χ^2_{16}=30.061, P=0.018), with group 5 having the most negative attitudes. The majority of respondents on Mbirikani (64%) claimed to like the presence of wild herbivores on their ranch, as compared with only 24% of Merueshi household heads. Forty-five percent of Mbirikani members professed to like living with carnivores, while only 3% of respondents on Merueshi group ranch (one person) said they liked having carnivores on their ranch.

The details of reasons given for liking or disliking herbivores and carnivores are given in Appendix 5A. Effectively, for Mbirikani, wildlife bursaries had the greatest positive influence on peoples' attitudes to herbivores, with the perception that herbivores attract tourists and create jobs as the next most important reason. For those who disliked herbivores, the spread of disease was the major reason given and resource competition and crop damage were the next most important reasons. Regarding carnivores, the perception that these species attract tourists and create jobs had the greatest influence on people's attitudes, followed by the presence of the Predator Compensation Fund (PCF). Education bursaries and conservation projects were also important reasons given for liking carnivores. Unsurprisingly, the main reasons given for disliking carnivores were that they kill and injure livestock and pose a threat to human life.

For Merueshi households, who received none of the financial benefits that Mbirikani did, the main reason given for liking herbivores was cropping (sustainable harvesting of certain herbivore species), with the hope that it would be reintroduced soon. The presence of wildlife bursaries was also mentioned, as they could see how this had benefited their neighbours. Competition for resources was the major determinant of negative attitudes to herbivores. This is consistent with their perception that resource competition was a major cause of conflict with wildlife. Disease transmission from wildlife also had an important influence on Merueshi households' attitudes. Only one person on Merueshi claimed to like carnivores, saying that they brought in tourists and were attractive to look at. Negative attitudes to carnivores were for the same reasons as Mbirikani; livestock death and injury, and a threat to human life.

However, grouping results by ranch may mask important regional differences. Chi-square analyses indicated that there were significant differences between groups in reasons given for liking both herbivores and carnivores (χ^2_{20}=59.82, P<0.001 and χ^2_{12}=22.62, P=0.031 respectively). There was also a significant difference between groups in reasons given for disliking herbivores (χ^2_{15}=77.10, P<0.001), but there was no difference in reasons for the dislike of carnivores (χ^2_{15}=18.61, P=0.232). Appendix 5A gives details of which reasons were most important to which groups. Effectively, well-educated groups close to major services mentioned the creation of jobs and tourism as major reasons for liking herbivores and carnivores whilst marginalised groups did not. The groups involved in crop farming mentioned, significantly more than would be expected by chance, that herbivores damaged crops and could injure people and livestock. Merueshi households reported resource competition as a reason for disliking herbivores significantly more than would be expected by chance.

5.3.4 Factors affecting attitudes towards wildlife

This section uses ordinal logistic regression analyses to investigate which factors played an important role in affecting households' overall attitudes to wildlife. Mbirikani and Merueshi Group Ranches were analysed independently. The independent variables used were from the questionnaire responses. Calculations of costs to wildlife were based on perceived losses and are described in detail in Chapter 4.

5.3.4.1 Mbirikani

An ordinal logistic regression was used to determine which variables affected a respondent's attitude to wildlife on Mbirikani Group Ranch. The dependent variable was the attitude score allocated to each respondent, from 1 (most negative) to 5 (most positive). The independent variables included are shown in Table 5.3. The overall result was significant (G_{22}=37.879, P=0.019), and the model had a good fit (log-likelihood = -199.326; Pearson χ^2_{566}= 590.531, P=0.230). The results are summarised in Table 5.3.

The results show that 'group' (effectively region of habitation) had a significant affect on attitudes, with group 5 being significantly more negative towards wildlife than all the other groups. The simple presence of any financial wildlife benefit was enough to improve people's attitudes to wildlife significantly, although the actual *amount* of money generated did not significantly affect attitudes. The scale of costs from wildlife (from whatever cause) did not affect people's attitudes. Perceived presence or absence of access to compensation for depredated livestock was the only other variable to have a significant affect on attitudes.

Table 5.3 Results of the ordinal regression for Mbirikani Group Ranch. CV=continuous variable, BV=binary variable, (y) = yes; indicates presence of the variable.

Independent variables	*Description of variables*	*Z*	*P*	*Odds ratio*
Group code (gp 5 = ref)				
1	Categorical variable; (Groups 1, 2 ,3 ,4 and 5 on Mbirikani)	-2.20	0.028*	0.27
2		-2.50	0.012*	0.21
3		-2.84	0.004*	0.16
4		-3.19	0.001**	0.13
Clan (clan 1 = ref)	Categorical variable; Clans on Mbirikani (1= Ilaiser, 2=Ilmolelian, 3=Ilaitayiok)			
2		-0.38	0.703	0.86
3		-0.23	0.815	0.81
Education	CV; number of years of education	-0.54	0.587	0.97
Income from wildlife (y)	BV; presence/absence of income from wildlife	-2.44	0.015*	0.38
Total wildlife benefits	CV; total financial income from wildlife (US$)	-0.08	0.939	1.00
Total cost from wildlife	CV; total cost from wildlife (US$)	-0.51	0.611	1.00
Cost to predation	CV; cost from predation (US$)	0.92	0.358	1.00
Cost to wildlife disease	CV; cost from wildlife-related disease (US$)	-0.30	0.766	1.00
Cost to ECF	CV; cost from east coast fever (US$)	0.58	0.562	1.00
Cost to MCF	CV; cost from malignant catarrhal fever (US$)	0.58	0.565	1.00
Cost to trypanosomiasis	CV; cost from trypanosomiasis	0.52	0.604	1.00
Household size	CV; number of people within the household	0.23	0.817	1.01
Total livestock holding	CV; total number of livestock owned	0.27	0.789	1.00
Shambas (y)	BV; presence/absence of a shamba	1.04	0.298	1.58
Employment (y)	BV; presence/absence of official employment	-1.08	0.281	0.61
Business (y)	BV; presence/absence of a business	-0.46	0.643	0.81
Compensation (y)	BV; presence/absence of compensation	-2.28	0.023*	0.06
Land ownership (y)	BV; presence/absence of title deeds to land	1.53	0.125	3.29

* = P<0.05, ** = P<0.01

5.3.4.2 Merueshi

The ordinal logistic regression for Merueshi Group Ranch included fewer independent variables than Mbirikani, but the dependent variable was the same (attitude score to wildlife). Income from wildlife and total wildlife benefits were excluded because every respondent reported zero. Likewise every Merueshi respondent owned land and did not receive compensation, so these variables were also excluded. Colinearity issues with 'total cost to wildlife disease' prevented cost of ECF being included in the model.

The overall result of the model was significant (G_{13}=54.211, P<0.001), and the model had a good fit (log-likelihood = -6.356; Pearson χ^2_{71}= 11.206, P=1.000). However no variables appeared as significant in affecting attitudes to wildlife. The results are presented in Table 5.4. The explanation of the variables is as in Table 5.3.

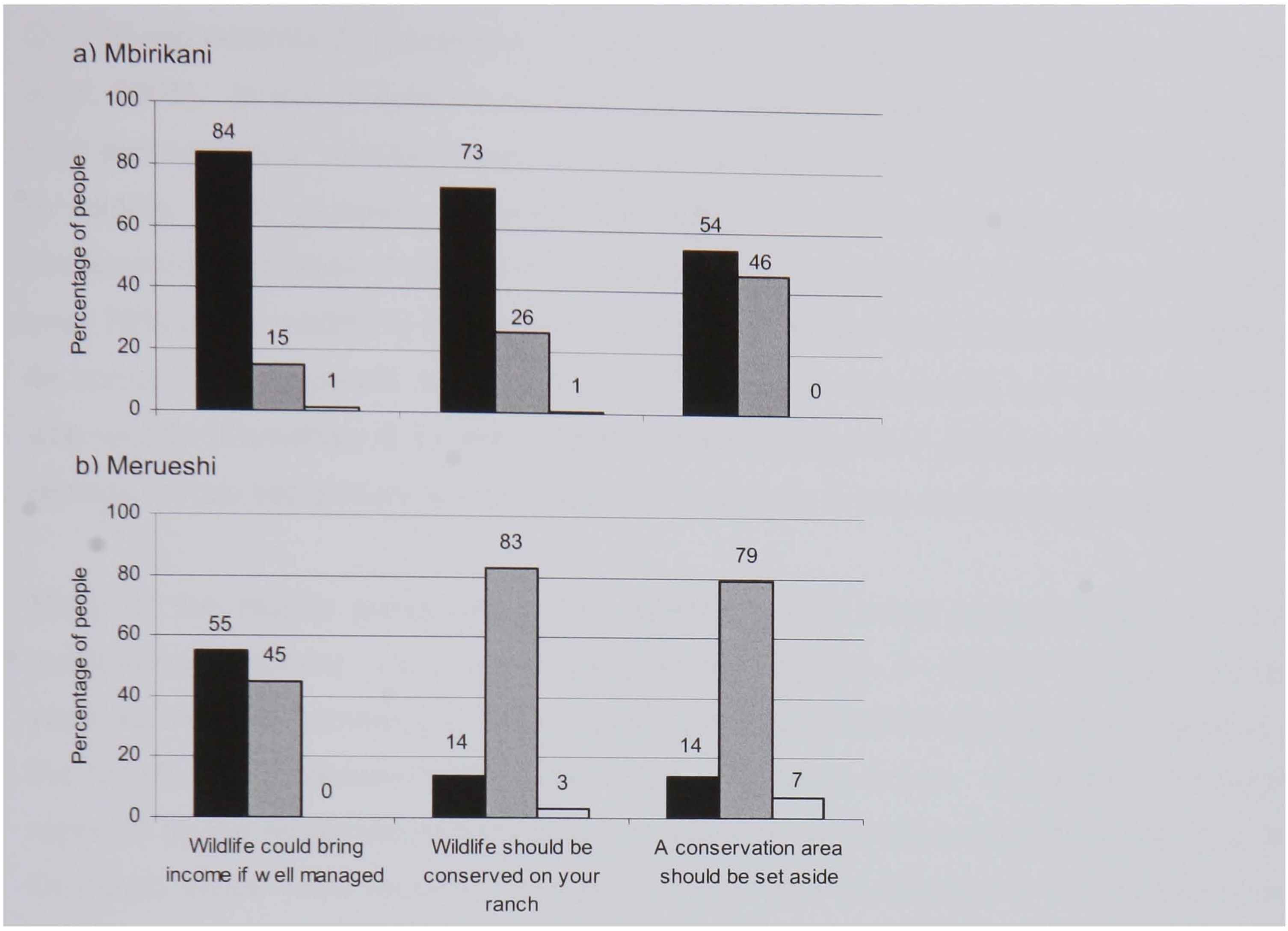
Table 5.4 Results of the ordinal logistic regression for Merueshi Group Ranch

Independent variables	*Z*	*P*	*Odds ratio*
Clan (clan 1 = ref)			
2	-1.43	0.154	0.00
3	1.25	0.213	2.05E+10
Education	-1.18	0.240	0.30
Total cost from wildlife	-1.40	0.163	0.99
Total cost to predation	1.42	0.157	1.01
Total cost to wildlife disease	1.40	0.161	1.01
Total cost to MCF	-1.49	0.137	1.00
Total cost to nagana	-1.30	0.194	1.00
Household size	-1.28	0.202	0.20
Total livestock holding	1.17	0.240	1.41
Shambas (y)	-1.47	0.143	0.00
Employment (y)	-1.34	0.180	0.00
Business (y)	-1.45	0.146	0.00

5.3.5 Attitudes towards conservation and indicators of behavioural change

Attitudes towards conservation were determined by responses to a series of statements which required the respondent to agree or disagree. These results are illustrated in Figure 5.2 and clearly demonstrate that Mbirikani respondents are far more positive towards wildlife conservation than Merueshi. Over 70% of Mbirikani households were in favour of conserving wildlife on their ranch, and 54% agreed with the idea of setting aside a conservation area for wildlife on their land. Only 14% of Merueshi respondents agreed with either of these.

Figure 5.2 Responses to conservation statements by a) Mbirikani households (N=148) and b) Merueshi households (N=29) expressed as percentages. Black = agree, grey = disagree, white = don't know.



Behavioural change is very difficult to measure in a short term study. A willingness to reduce livestock in favour of wildlife may provide a measure of potential behavioural shifts in the future. However, even if wildlife were to be very profitable, it is still not easy to convince the Maasai to reduce livestock in favour of wildlife. When asked where they would like their income to come from, if wildlife revenues equalled livestock revenues, by far the majority of respondents (82% on Mbirikani and 72% on Merueshi) wanted to retain livestock as their main source of income. When asked if they would decrease livestock herds to encourage more wildlife, if wildlife were to be *more* profitable than livestock, only 34% on Mbirikani and 62% on Merueshi said yes. A willingness to leave personal land unfenced for the benefit of wildlife can provide another indicator of pro-wildlife behaviour. Nonetheless, 53% of Mbirikani and 76% of Merueshi households said they *would* fence their land specifically to keep wildlife away.

However, recent analysis of the rate of lion killings on Mbirikani Group Ranch has shown a decline in the number of lions killed on Mbirikani from at least 24 in 2002 to 1 in 2004 and 2 in each of 2005 and 2006 (Hazzah, Maclennan & Frank 2007), which suggests a possible behavioural change in a pro-conservation direction.

5.4 DISCUSSION

Quantifying community perceptions is key in translating ecology into management (White *et al.* 2005). In this chapter, focus is on the Maasai pastoralists of southern Kenya, and their perceptions of wildlife related costs and benefits, which in turn affected their attitudes to wildlife. It is generally believed that local communities are more likely to support conservation initiatives if they receive direct benefits from them (McNeely 1995). Since over 75% of the wildlife in Kenya lies outside of National Parks (Ottichilo *et al.* 2000) and its conservation depends on the activities of the local landowners and their compatibility with wildlife (Earnshaw & Emerton 2000), understanding their attitudes and willingness to change certain behaviours is an important prerequisite to any management policy.

Many of the results presented in this chapter, which were gathered during a formal questionnaire survey, were consistently re-emphasized in informal group discussion sessions with the community. This unprompted support of the findings gives credibility to the results and increases confidence in the conclusions drawn. In addition, the benefits reported by the respondents corresponded well with what was known to be the case from Ol Donyo Wuas Trust records. The perceived problems reported in the interviews were the same issues that have been reported informally for the three years in which I have been living in the area. Whilst there is always cause for concern of biases in interviews of this type (Hazzah 2007), my relationship with the community was built up over two years before the questionnaires were undertaken, creating a mutual degree of trust and cooperation. All these reasons give cause to believe that the results presented here represent the true attitudes and opinions of the households.

5.4.1 Wildlife benefits generated on Mbirikani Group Ranch

Table 5.1 shows that in 2005, Mbirikani Group Ranch generated approximately \$230,450 or \$2.1 per acre from wildlife. Norton Griffiths *et al.* (in press) estimated the mean wildlife rents received by landowners in Kenya to be \$1.6 per acre per year, with 95% confidence limits of between \$0.9 and \$2.7 per acre per year, which is consistent with results from Mbirikani. The same study estimated that for an area such as Mbirikani, with a mean annual rainfall of 350mm (see Altmann *et al.* 2002), an income from wildlife of \$4.5 per acre per year represents the threshold value at which a mixed wildlife-livestock farming system becomes the optimal land use (Norton-Griffiths *et al.* in press). Below this critical level, it would be in the landowners' best interests to rid their land of wildlife. This indicates that wildlife revenues on Mbirikani need to double in order to give wildlife a chance in an economically competitive future. All the same, the extent of revenue sharing by Ol Donyo Wuas Lodge (21%) is high in comparison to most of the country. In general in Kenya, landowners find it difficult to capture more than 5-10% of the wildlife rents (Norton-Griffiths & Butt 2006), less than half of what Mbirikani members are receiving

Of the total $230,450 generated for Mbirikani Group Ranch in 2005, over $48,000 (21%) was paid directly to the group ranch committee in the form of conservation fees, land rents, and research fees. In theory this should have been used for the benefit of the whole community. However, as the results illustrated, very little if any, of what was paid to the committee reached the rest of the community. This misappropriation of funds is a familiar story over much of Maasailand. In an example cited in Thompson & Homewood (2002), only 16% of a wildlife association's income (paid through a committee) was received by the group ranch members in one year, whilst the next year nothing was received. This results in only a few members benefiting significantly from wildlife earnings (Ogutu 2002; Thompson & Homewood 2002). These 'empowered few' help to create awareness among the rest of the members to participate in ecotourism initiatives, whilst at the same time marginalising them in benefiting from the wildlife revenues generated (Ogutu 2002). This was clearly the case on Mbirikani.

5.4.1.1 Spatial distribution of benefits

Results indicate that benefits were not evenly distributed around the ranch. Group 2 households received the most benefits from wildlife, followed by group 3, the two groups which were closest to towns, schools, main roads and trading centres. These were also the two groups with the most educated household heads and home to the members of the group ranch committee. Group 4 received the least benefits from wildlife: a few members were employed as game scouts and they were all entitled to compensation but otherwise no benefits were generated by that community. During feedback workshops, members of groups 4 and 5 repeatedly complained about the uneven share of benefits and were angry at the committee's misappropriation of funds meant to benefit them. Not only did these groups receive the least benefit from wildlife, they also suffered the highest costs (see Table 4.3, Chapter 4). Furthermore, only 24% of households on Mbirikani received any financial benefits from wildlife and Figure 5.1 illustrates the enormous inequality in distribution of revenues. Such disparity in income can engender negative feelings towards the wildlife resource due to the perception of inequitable sharing of the revenue (Hazzah 2007).

The inequitable distribution of benefits is also one of the main drivers of land privatisation. People felt this would decrease inequality between rich and poor by providing everyone with an equal share in land, and would allow households to capture benefits at an individual level, rather than through an institution such as the group ranch committee (Norton-Griffiths *et al.* in press).

5.4.2 Perceptions of problems associated with wildlife

Peoples' perceptions of conflict with wildlife play a considerable role in shaping their attitudes towards wildlife conservation (Marker *et al.* 2003; Gadd 2005). Different areas may suffer different problems with wildlife, as illustrated for Mbirikani and Merueshi. For example, the perception of resource competition was much greater for Merueshi households. Since Merueshi was subdivided, the natural resources such as grass and water were privately owned, creating more of an opportunity for personal conflict with wild herbivores. In addition, the majority of land on Merueshi was degraded and overgrazed; ground cover by grass was significantly lower than on Mbirikani (Chapter 2). It is easy to see therefore why pastoralists who are already struggling to find sufficient grazing within their subdivided ranch would perceive a much greater level of conflict over resources with wildlife than Mbirikani households. All such inter-regional differences should be taken into account when trying to understand attitudes towards wildlife and during the formation of any conflict-mitigation strategies.

5.4.3 Attitudes to herbivores and carnivores

Both benefits received by wildlife and the perception of problems and costs from wildlife play a part in shaping peoples' attitudes towards wildlife conservation. The term wildlife however is a sweeping term covering a wide array of species with different types of interactions with humans and their livestock. It was therefore considered important to investigate households' attitudes to herbivores and carnivores separately.

Unsurprisingly, carnivores were less popular than herbivores on both ranches. Of those people that claimed to like herbivores (64% on Mbirikani and 24% on Merueshi), reasons given were mostly benefit-related, especially education bursaries and employment opportunities (see Appendix 5A). Positive attitudes to carnivores (45% on Mbirikani and 3% on Merueshi) were once again benefit-related. Carnivores, more so than herbivores, were perceived to be responsible for attracting tourists and creating jobs. However, one of the major factors resulting in a positive attitude to carnivores (for Mbirikani members) was compensation. This is because compensation offsets some of the financial loss of a livestock death to carnivores, which previously the herd owner would have had to bear on his own. Nonetheless it is noteworthy that the presence of the PCF caused so many people to claim to like carnivores, given that the amount of compensation paid was on average considerably less than the market value of the livestock killed (S. Maclennan, unpublished data). Thus carnivores were still causing the household a financial loss, as well as an emotional one.

Intuitively it seems a good thing that tourism and conservation efforts bring wildlife revenues to communities and create positive attitudes. However, if the motivation to conserve wildlife becomes purely financial, with aesthetic or cultural benefits forgotten, there may be a major problem if the financial incentives are lost (Gadd 2005). This is a potential problem with the compensation project, especially given the importance of this in shaping people's attitudes to carnivores. Indeed in this study, many people said that they *only* liked carnivores because of the PCF, and if that were not present they would have no reason to like them. This demonstrates the power of PCF, but also highlights the importance of ensuring long-term sustainability of such a project. It would also be advantageous to conservation efforts in the future to try to encourage local cultural values, and non-financial conservation motives within local communities (Gadd 2005), especially due to the potential un-sustainability of compensation, and the fickle and volatile nature of tourism (Walpole & Leader-Williams 2002).

Despite compensation and other benefits from wildlife experienced by Mbirikani members, there was still a considerable proportion of household heads who reported to dislike both herbivores and carnivores. The vast majority of Merueshi households disliked both. The reasons for disliking carnivores (livestock depredation and injury to both livestock and people) were straightforward and widespread and there were no significant differences between groups in the reasons given. Reasons for disliking herbivores, however, differed significantly between groups, but were mostly related to the spread of wildlife disease and the perception of competition for resources. The perception of resource competition merits further attention. According to some scientists, there is little evidence that livestock numbers are reduced by high densities of wildlife (Prins 2000), and where competitive interactions do occur their costs are described as 'negligible' (Deodatus 2000; Prins 2000). On the other hand, Norton-Griffiths *et al.* (in press) present a case study from the Maasai Mara where they estimate that net returns to livestock would be 48% higher if wildlife were eliminated. There is no evidence for Mbirikani and Merueshi as to whether or not resource competition is actually occurring on the scale perceived by pastoralists. Nonetheless, as with most issues of conflict, it is the perception of a problem which drive attitudes and actions (Mishra 1997), so whether real or imagined, the issue of conflict for resources must be considered important. It is noteworthy that the two groups (4 and 5) with easy access to grazing in the Chyulu Hills National Park, report problems with resource competition significantly less often than expected by chance, whilst those in more overgrazed areas report it more.

It is very clear from the results in Section 5.3.3 that, whilst there may be a widespread like or dislike of wildlife, the reasons vary significantly spatially. An understanding of this is important when trying to develop conflict resolution strategies to encourage tolerance. Different regions may require different approaches (e.g. see Weladji *et al.* 2003). For instance, it may be sensible to distribute benefits between areas in proportion to the attitudes of the people and their perceived level of conflict with wildlife. The type of benefit could also be specifically targeted according to the reasons people give for disliking wildlife. For example where carnivores are hated, predator compensation may be a worthwhile intervention strategy, whereas where the main problem is with resource competition, educational forums and more general benefits like education bursaries and employment might be more effective.

5.4.4 Factors affecting attitudes to wildlife

In the light of the above, understanding which factors influence attitudes and tolerance towards wildlife is critical for choosing the most appropriate solutions to conflict (Zimmerman, Walpole & Leader-Williams 2005). These may be mitigations to reduce losses (Ogada *et al.* 2003), education campaigns to improve awareness (Marker *et al.* 2003; Weladji *et al.* 2003) or the generation of financial incentives (Mishra *et al.* 2003; Bulte & Rondeau 2005). The results of the ordinal logistic regression analyses presented in Section 5.3.4 illustrate the importance of financial benefits in improving attitudes towards wildlife and highlight the value of equitable distribution of these benefits.

5.4.4.1 Mbirikani

On Mbirikani Group Ranch, the binary variables of presence or absence of wildlife benefits and compensation significantly affected people's attitudes to wildlife. In both cases, 'presence' increased the likelihood of a higher (more positive) score for attitude to wildlife. In addition, 'group' had a significant effect, such that respondents in group 5 (the reference group) were significantly more likely to have a negative attitude to wildlife than respondents in all other groups. These are discussed in turn below.

The simple presence of any wildlife benefits was enough to significantly affect people's attitudes to wildlife in a positive way ($Z=-2.44$, $P=0.015$). However the *amount* of revenue received by the household from wildlife ('total wildlife benefits') had no significant effect on people's attitudes. Likewise, the amount of money lost by the household as a result of living with wildlife ('total cost from wildlife') had no significant effect on people's attitudes. Even when broken down by predation and different wildlife-related diseases, the scale of these costs did not affect people's attitudes. Other studies have found similar results. Weladji *et al.* (2003) found that local peoples' attitudes towards the Bénoué Wildlife Conservation Area in Cameroon were not significantly affected by the extent of wildlife damage. Heinen (1993) found the same in Nepal, as did Fiallo & Jacobson (1995) in Ecuador.

This has interesting implications for management. The results suggest that conservation efforts should focus on increasing the *spread* of wildlife benefits more than simply aiming to increase the total revenue generated, and should prioritise increasing the benefits over decreasing the costs. Both these points were emphasised in the feedback workshops where people in marginalised groups consistently complained about the 'unfair distribution' of wildlife benefits, while very little was mentioned about revenues being too low. Whilst there were certainly complaints about the costs from wildlife (mostly predation), this was always related to the concurrent lack of benefits in the most problematic areas. In addition, community awareness of the current scale and distribution of wildlife benefits is very important. In the latest community discussion sessions, local Maasai were astounded to hear how much money was generated through wildlife; they had no idea of the magnitude of the benefits. Even something as simple as ensuring the wildlife revenues generated are well publicised is likely to have a considerable affect on people's attitudes.

Wildlife revenues do not have to benefit every household for them to serve the purpose of improving attitudes. Studies have found that even having a close friend or relative who benefits from wildlife is sufficient to engender a more positive attitude to wildlife (J. Roque de Pinho, unpublished data). Indeed this study found that double the number of people claimed to like herbivores or carnivores because they perceived them as bringing educational bursaries to others (20%), than those who liked them due to receiving bursaries personally (10%). This is encouraging as it suggests that not every single person needs to benefit personally from wildlife in order to see its value, but does emphasise the importance of distributing benefits widely, and perhaps even deliberately trying to ensure that at least one member of every extended family receives some benefit from wildlife.

The ability to receive compensation was the second variable which the ordinal logistic regression found to be significant in affecting people's attitudes to wildlife. On Mbirikani Group Ranch, compensation was available to everyone. At the time of the interviews however, there were some disputes over the compensation system and several people from higher conflict areas had decided to veto the scheme, thus answering 'no' when asked whether they would be entitled to compensation in the event of depredation on their livestock. These people were generally very disillusioned with wildlife and conservation efforts (see Hazzah 2007), and so it is unsurprising that they had significantly lower attitude to wildlife scores than those who answered yes to the compensation question. The results for the rest of the model did not change if this variable was removed.

The final variable found to significantly affect peoples' attitudes to wildlife was group, i.e. region of habitation. Group 5 was shown to have significantly more negative attitudes to wildlife than all the other groups. This is due to several reasons. Perception of losses showed group 5 to suffer most from both predation and wildlife-related diseases (Chapter 4). In addition, they received fewer benefits than some other groups, which was a source of friction (see Table 5.2), especially since the benefits they did receive were disproportionately low in relation to the conflict they endured (Hazzah 2007). This group has also been described as politically and socially marginalised by both conservation initiatives and their own ranch, which incites negative attitudes (Hazzah 2007).

Taking livestock holding as an indicator of wealth, it is important to note there was no significant effect of wealth on people's attitudes to wildlife. This is consistent with other studies, which use 'household income' as an indicator of wealth (e.g. Parry & Campbell

1992; Heinen 1993; Weladji *et al.* 2003). Employment, business, household size, shamba ownership and land ownership were also statistically unimportant in affecting attitudes to wildlife, as was the level of education of the respondent. Many other authors report no significant effect of education on local peoples' attitudes to wildlife and conservation (e.g. Parry & Campbell 1992; Newmark *et al.* 1993; Weladji *et al.* 2003), although some studies did find it a significantly important variable (e.g. Heinen 1993; Fiallo & Jacobson 1995). Infield & Namara (2001) working in Uganda, found that attitudes were significantly influenced by land ownership, which is contrary to the findings in this study. This is most likely due to the near-uniformity in land ownership in this study; only 9 households (6%) on Mbirikani owned land.

5.4.4.2 Merueshi

None of the variables included in the ordinal logistic regression for Merueshi Group Ranch were found to affect people's attitudes to wildlife significantly. This may be due to the fairly low variation of attitudes on Merueshi. Almost 50% of the respondents had a score of 1, whilst only one person scored 4. There were no scores of 5 (the most positive) on Merueshi. The sample size for Merueshi (n=29) was also fairly small, and there were fewer variables within the regression model, since no household received any benefits or compensation. It is interesting to note however that, as on Mbirikani, the scale of costs from wildlife did not significantly affect the attitude of the household head towards wildlife. Attitudes were uniformly poor, with no particular reasons responsible.

5.4.5 Attitudes towards conservation and indicators of behavioural change

Participants' responses to conservation-oriented statements (presented in Figure 5.2) illustrate the considerable difference in attitudes to wildlife conservation between Mbirikani and Merueshi. On Merueshi, the vast majority of people (83%) disagree that wildlife should be conserved on their ranch, as compared with only 26% on Mbirikani. This is despite the fact that 55% of Merueshi respondents agreed that wildlife could bring 'lots of income' to their households if it were managed properly. So although many Merueshi household heads recognised that wildlife could potentially be a valuable resource, very few (14%) showed any desire to try and conserve it. A much higher percentage of people on Mbirikani agreed that wildlife could potentially bring them considerable income (84%). This probably reflects the fact that many have already either received income personally or have witnessed others doing so. Whilst 73% of Mbirikani households agreed that wildlife should be conserved on the ranch, only 54% agreed that an area of the ranch should be set aside as a wildlife conservation area. This suggests that members may have other suggestions for the conservation of their wildlife, and merits further investigation.

Whilst attitudes to conservation were fairly positive on Mbirikani, several indicators of behavioural change suggest that people are not yet willing to change their lifestyles to accommodate wildlife. It is generally understood that fencing small parcels of land can have a negative impact on wildlife conservation (Boone & Hobbs 2004), and yet 53% of Mbirikani households agreed that all landowners should fence their land to keep wildlife out. This suggests that people may see wildlife conservation in an isolated sense, i.e. 'it can be conserved somewhere on the ranch, but not on my land', and suggests the need for further focussed educational efforts about the underlying requirements for wildlife conservation such as freedom of movement and access to heterogeneous resources.

A willingness to reduce livestock in favour of wildlife could also be considered an indicator of potential behavioural change, although this is an extreme test for Maasai pastoralists. Results showed that even if wildlife returns were to be higher than livestock returns, only 34% of Mbirikani and 62% of Merueshi household heads expressed willingness to decrease their livestock herds to encourage wildlife. However, decreasing livestock herds to favour 'more-profitable' wildlife would be far more than just an economic decision, as the Maasai pastoral identity is closely tied with their livestock herds which are an important indicator of their social status (Spear 1993). Moreover, these were hypothetical situations, posed in a 'what if' fashion, and probably do not accurately reflect what would happen if this were to become reality, and money from wildlife became a considerable income. Nonetheless, it is noteworthy that Merueshi households, rather than Mbirikani ones, reported greater willingness to decrease livestock herds in favour of wildlife, despite the fact that Mbirikani households experienced the benefits wildlife can bring. This may be because Merueshi households struggle with livestock farming due to their heavily overgrazed land (Chapter 2), and may see wildlife as an easy option for generating income. However, they had little wildlife on their ranch (Chapter 2). In contrast, Mbirikani had a reasonable amount of wildlife and many respondents saw it as the source of numerous problems; thus they were less willing to encourage more of it.

The apparent behavioural change suggested by the decrease in lion killings may represent a case where wildlife incentives have resulted in behavioural change in a pro-conservation direction. However, this reduction in lion killings could be due to a variety of factors, including the conservation efforts on the ranch by the PCF and Kilimanjaro Lion Conservation Project, or simply the difficulty in finding and killing the few remaining lions (S. Maclennan, *pers. comm.*).

5.4.6 Conclusion

Evidence presented in this chapter suggests that financial incentives from wildlife can improve community attitudes towards both wildlife and its conservation. However, it is the *distribution* of benefits which emerges as most influential in shaping attitudes, more important even than the amount of money provided, or the costs incurred from wildlife. Specific feelings towards wildlife were influenced by perceptions of conflict, and these differed significantly between regions. Where no wildlife benefits were currently available, attitudes to conservation were significantly more negative than where revenue had been generated, even with the enormous inequality of the revenue sharing. Despite the willingness to conserve wildlife on Mbirikani, there was little evidence that people were willing to change their behaviour to promote it, with the possible exception of the reduction in the killing of lions.

In summary therefore, evidence is presented to support the hypothesis that that wildlife revenues can positively influence pastoralists' attitudes to wildlife but are currently insufficient to create behavioural change. A more equitable distribution of earnings from wildlife would almost certainly considerably increase support for wildlife, and educational efforts are important for people to understand how conservation works.

This chapter has concluded the data chapters of the thesis by taking a community perspective on the issues of wildlife conservation. The following, final chapter draws together information presented in this and the preceding three chapters to formulate a conservation plan for Mbirikani.

CHAPTER 6
THESIS SUMMARY AND DISCUSSION

6.1 INTRODUCTION

The original aim of this multidisciplinary thesis was to investigate the threats to wildlife conservation posed by land subdivision and sedentarisation of the Kenyan Maasai, as well as the potential for conservation in terms of the underlying economic context and local attitudes. The main goal was to produce a management plan for the group ranches of the Amboseli-Tsavo Ecosystem as land tenure policies change. To this end, the following chapter presents a conservation plan for Mbirikani Group Ranch, while this chapter summarises the main findings of the thesis, discusses their implications within a global context and presents recommendations for future research.

6.1.1 Challenges facing wildlife conservation and management in Kenya

Wildlife is declining in Kenya at a rate of 3-4% per annum (Kock *et al.* 2002). Since hunting was banned in Kenya, 60-70% of the country's wildlife has been extirpated (Norton-Griffiths 2007). Kenya's wildlife continues to be threatened by a host of environmental and political impacts, most of which stem from the burgeoning human population growth; 5.54% per annum in Kenya's Kajiado District (Republic of Kenya 1997). Major direct threats include deforestation and habitat clearance, over-utilisation of rangelands by livestock resulting in degradation of environmental resources, illegal exploitation of plant and animal resources, increasing human-wildlife conflict, unregulated agricultural enterprises and industrial pollution (Norton-Griffiths 1996; Kiringe, Okello & Ekajul 2007). The ability for local people to receive worthwhile revenues from wildlife is prevented by the ban on all consumptive use of wildlife (Norton-Griffiths 2007), making the generation of sufficient economic incentives to conserve wildlife a difficult, albeit critical, target.

Only 7-8% of Kenya's total land area is under formal protection (Kinyua *et al.* 2000) and over 70% of the country's wildlife lives outside these parks and reserves (Grunblatt *et al.* 1995a), in Kenya's arid to semi-arid rangelands. This means private landowners and communal property stakeholders have the greatest potential to sustainably manage wildlife. However, political instability (Fratkin & Roth 2005), immense poverty, a deep rooted cultural involvement with livestock and environmental challenges such as drought (Campbell 1999) make wildlife conservation a distant priority for the inhabitants of these rangelands. Conservation is therefore a challenge which is just as much about the people as it is the wildlife.

With this in mind, this thesis has aimed to provide a framework for implementing community based natural resource management in the Maasai lands in southern Kenya. While many of the lessons learned from this thesis are applicable throughout the developing world (and examples are mentioned throughout this discussion), it is important to remember that ecosystems, cultures, socio-economic levels, land tenure and political systems differ markedly throughout the world, ultimately requiring localized approaches and strategies.

To investigate threats to conservation and possible intervention strategies, one requires a comprehensive, holistic approach to the issues involved (Barrett & Grizzle 1999). In the case of the threat posed by land subdivision and pastoral sedentarisation to the integrity of the Amboseli-Tsavo Ecosystem, a holistic approach required the inclusion of ecological, economic and social aspects to the research. For example, a conclusion from an ecological study that subdivision was a potential threat to the environment and the wildlife, combined with suggestions for how to mitigate this problem, would be only half the picture. The Amboseli-Tsavo Ecosystem is home to over 36,000 people (Croze *et al.* 2006), whose views and opinions are of the utmost importance in determining the management of their lands. These attitudes are likely to be influenced, at least in part, by the economic context in which they live, making essential a comprehensive investigation of the economic situation at a household level. The importance of this holistic approach to conservation is becoming increasingly recognised (Gilmore 1997; Barrett & Grizzle 1999; Banks 2004) and should be applied in all situations where conservation efforts require the co-habitation of local people and wildlife. Indeed in Mexico, an ambitious wildlife and conservation management initiative aims, as its main objective, to *"integrate environmental, economic, social and legal strategies to address wildlife needs while promoting broader societal participation"* (Valdez *et al.* 2006).

This thesis has tackled all these issues using a case study from two Maasai ranches in southern Kenya, located within critical wildlife habitat. The following section will summarise the main results from the thesis, followed by a discussion of the relevance of these findings to other conservation initiatives around the world.

6.1.2 Summary of main findings

The ecological results presented in Chapter 2 illustrate that land subdivision and the consequent sedentarisation of pastoralists can dramatically decrease the wildlife populations, as well as the grazing resource. Having been at a similar level in the 1970s and early 1980s, by 2005 wildlife densities on subdivided Merueshi were considerably and significantly lower than on Mbirikani. This supports findings by several other authors (Norton-Griffiths 1998; Seno & Shaw 2002; Worden *et al.* 2003) which mention the loss of wildlife on subdivided land parcels. Likewise there was significantly less grass on Merueshi in 2005 and the remaining grass resource was under consistently higher pressure from herbivore grazing. Both are well-documented effects of pastoral sedentarisation (Salzman 1980; Boone 2005; Leloup 2006) where livestock is restricted to certain patches throughout the year and the grazed grass is given no time to recover.

Another recognised consequence of subdivision and sedentarisation, a decreased freedom of movement, is reflected in the distribution of both permanent and temporary bomas on the two ranches. On Mbirikani, permanent bomas were clustered around permanent water sources with a considerable number of temporary bomas spread throughout the ranch in both dry and wet seasons. On Merueshi, the majority of permanent bomas were randomly distributed around the ranch and there were rarely any temporary bomas on the ranch. The ability to disperse or congregate according to season reflects a freedom of movement and open access to widely dispersed resources (Schwartz 2005) and may be a useful indicator of the degree of sedentarisation in pastoral communities.

Despite the obvious impact on the environment of subdivision and sedentarisation, results from Chapter 3 suggest these processes had little effect on the distribution patterns of wild grazers. Whilst this may actually be the case, it may also be simply due to the resource-stressed environment in which the study was carried out, whereby all grazers were forced to congregate in areas where there was some grass available, irrespective of other constraints. This tentatively suggests that in areas of very low vegetation productivity, land subdivision and decreasing freedom of movement, has less effect on wildlife distributions than it would if a ranch of high productivity were to be subdivided. This is supported by Boone *et al.* (2005) who demonstrated that the importance of subdivision relative to herbivore carrying capacity was weak for areas of very low (or very high) vegetation productivity, but that subdivision had a much greater impact on ranches of intermediate productivity (Boone *et al.* 2005). However, as discussed in Chapter 3, the analysis could have been considerably improved and until further investigation is carried out into this issue, these results should be treated as preliminary. Interestingly, Ash, Gross & Smith (2004) review the issues of scale and landscape heterogeneity on herbivore production and distribution patterns, and state that at intermediate scales (500-20,000ha), it has proved very difficult to predict the spatial distribution of grazers.

Chapter 4 took an economic angle and presents an analysis of the costs and benefits of wildlife to the Maasai pastoralists inhabiting Mbirikani and Merueshi Group Ranches. An assessment of the economic context in which conservation interventions are being proposed is of critical importance (Norton-Griffiths 1996; Norton-Griffiths 2006). Realistic economic models require accurate measurements of both the magnitude and distribution of costs to people who share their land with wildlife (Sutton *et al.* 2004), as well as quantification of benefits. Quantifying revenue generated however, is very different from quantifying the revenue actually received by the households due to corruption by local elites (Thompson & Homewood 2002). To do both, as in this study, is unusual, but I propose that it is of great relevance to understanding the extent to which future revenues will impact the ordinary community member.

The main findings were that wildlife-related disease and predation alone cost households on average $585 per year and that benefits, even at an average of $190 per household on Mbirikani, were far from sufficient to offset these costs. Despite the private land ownership and lower wildlife densities on Merueshi, wildlife-related costs were not significantly lower than on Mbirikani and nor was the overall deficit cost. Within Mbirikani however, there was a significant difference in wildlife-related costs between groups, with one group making, on average, a small profit from wildlife, while households in another lost almost $1000 from wildlife in 2005. A brief examination of the potential costs of grazing competition from wildlife indicated that these costs could be considerable, and a more comprehensive investigation into these costs would be worthwhile.

The final data chapter (Chapter 5) investigated the local people's attitudes towards wildlife and its conservation in relation to the wildlife revenues available. Such attitudinal surveys are important tools during all stages of community based conservation programs (Gillingham & Lee 1999). Significant findings include the fact that the wildlife revenues generated on Mbirikani were sufficient to engender significantly more positive attitudes to wildlife than for households on Merueshi, and moreover that it was simply the presence of wildlife benefits, not the scale, that significantly improved attitudes. Whilst revenue sharing on Mbirikani was good in comparison with most of Kenya (see Norton-Griffiths *et al.* in press), still only 24% of the group ranch members received any benefits (with the exception of predator compensation which was available to everyone), highlighting the importance of reducing inequalities and increasing the spread of wildlife benefits. This latter point was one of the major findings of this study. Whilst many conservation efforts focus on increasing the magnitude of the revenue available to local communities (e.g. Walton *et al.* 2006; Rowat & Engelhardt 2007), I propose that, whilst increasing revenues is certainly very important, a considerable improvement in attitudes could be generated at little extra cost by simply increasing the spread of current benefits. Additionally, simply increasing the amount of revenue available could actually engender more negative attitudes towards wildlife if the distribution is perceived to be inequitable (see Hazzah 2007). Nonetheless, the majority of Mbirikani members recognised that wildlife could bring considerable income to their households if it were to be properly managed, and were in favour of its conservation.

In summary, the ecological and socio-economic results suggest that wildlife conservation on Mbirikani is both necessary and feasible, but to accomplish it will require a considerable focus on improving the current economic situation.

6.1.3 Implications of results within the broader context

This thesis has implications relevant not only to the conservation of Mbirikani Group Ranch, but to the conservation of wildlife around the world where land use changes are occurring in the buffer areas of national parks and other protected areas.

The use of private lands adjacent to protected areas can have a major impact on the success or failure of the park's efforts to maintain regional biodiversity (Gosnell, Haggerty & Travis 2006), and particularly important is how the owners respond to conservation goals (Gosnell *et al.* 2006). In this thesis, I have highlighted how, in relatively poor pastoral areas with good potential for conservation, local people can be pro-conservation if benefits are present and equitably distributed.

The threat of physical subdivision of the land, a major conservation challenge in the Amboseli-Tsavo Ecosystem, is occurring (or has occurred) throughout the world, in all types of ecosystems and for many different reasons, and consequently this thesis may have global relevance. In the United States, for example, subdivision for housing is a major threat in the buffer zone of Yellowstone National Park (Gosnell *et al.* 2006), and in parts of India, the land holdings are reduced by half every 20-30 years due to subdivision between sons based on the succession laws (Anantha Ram *et al.* 1999). In almost every example, similar negative ecological consequences are reported; environmental degradation, decreasing biodiversity and reduced inefficiency of natural resource management. The subdivisions in the Indian case study, for example, resulted in a reduction of yields and a consequent shortfall of food on small farms, a decrease of soil fertility caused by continuous cultivation, and increasing desertification (Anantha Ram *et al.* 1999). Likewise, Knight, Wallace & Riebsame (1995) revealed negative ecological implications of subdivisions in the Colorado Mountains, including encroachment of development on fragile riparian areas and at forest edges.

Other major implications of land subdivision include an increase in fences, buildings and roads (Knight *et al.* 1995), and an increased fragmentation of the landscape, all of which are applicable in this Kenyan case study. These factors increase disturbance levels, which can disrupt both inter- and intra-specific wildlife interactions (Pomerantz *et al.* 1988) potentially resulting in altered wildlife communities (e.g. Skagen, Knight & Orians 1991). In addition, fences may truncate wildlife migratory routes and remove critical forage resources needed by wild herbivores (Boone & Hobbs 2004) leading to a reduction in overall population numbers. This latter statement is supported by results presented in this thesis, which highlight the necessity for conservation managers to try to minimise fences in rural areas. Additionally, the privatisation of single-owner land parcels has been considered the primary cause of forest fragmentation in Connecticut, USA over the past 40 years (Holdt, Civco & Hurd 2004). As with the situation in Kenya's rangelands, as parcel size decreases, and the number of land owners increases, the management of the forest resource becomes increasingly difficult (Holdt *et al.* 2004).

Despite the fairly ubiquitous, and potentially severe consequences of land subdivision, Knight *et al.* (1995) suggest that data required to evaluate the relationships between biodiversity and certain land use patterns is lacking. For example, more research is considered necessary to determine the relationship between certain species of wildlife and changes in housing density and distribution, in order for development to be compatible with wildlife conservation (Vogel 1989). Quantified data on the effects of land subdivision to wildlife populations is therefore of immediate and practical importance, and lessons learned from this Kenyan study may have wide-ranging relevance.

One further implication of land subdivision in pastoral areas is the concurrent decrease in mobility of the people and livestock leading to an increase in permanent settlements: sedentarisation. This process is taking place throughout the world (McPeak & Little 2005) and chapter 1 discussed several other factors which may cause or contribute towards sedentarisation. These included push factors such as certain government policies, loss of land, population growth, increasing social insecurities and the aftermath of drought and famine, as well as pull factors such as improved economic opportunities and access to infrastructure including markets and schools. Whatever the driving force however, where pastoralists settle in dry, marginal areas, frequently there are negative implications. This thesis adds to the body of literature describing negative environmental implications of pastoral sedentarisation (Darling & Farver 1972; Salzman 1980; Roth & Fratkin 2005), specifically a reduction in the quantity of grass available and the removal of water points for use by wildlife. Further investigation into the effect of sedentarisation on the quality of the grazing resource would be valuable.

Clearly subdivision and sedentarisation can be problematic for conservation efforts. However, the underlying process of land privatisation, even without physical division of the land or increased pastoral settlement, can also constitute a conservation challenge. This is because even in landscapes that look intact, underlying parcelization patterns reveal varying levels of ownership fragmentation (Walker *et al.* 2003 in Gosnell *et al.* 2006) which presents a challenge to conservationists due to the increasingly wide-ranging set of values, motivations and economic circumstances of the land owners (Gosnell *et al.* 2006). In the United States, for example, new land tenure patterns appear to be introducing new land use values to certain working landscapes (Walker & Fortmann 2003) which can affect the conservation potential. It is therefore important to consider the potential threats to conservation resulting from land privatisation, even if this does not entail fencing or sedentarisation.

In light of the above, it can be seen that land privatisation and subdivision and pastoral sedentarisation pose a conservation threat wherever they occur. This threat is imminent on Mbirikani Group Ranch and considering its location within one of the most biologically diverse landscapes in the world (Little 1996; Boitani *et. al.* 1998; Du Toit & Cumming 1999), the development of a conservation-based management plan should be a major priority. The following section discusses the current trends in conservation planning to provide a background to the conservation plan for Mbirikani presented in Chapter 7.

6.2 CONSERVATION PLANNING

Conservation planning in this case concerns *'the location and design of reserves that both represent the biodiversity of a region and enable the persistence of that biodiversity by maintaining key ecological and evolutionary processes'* (Desmet *et al.* 2002). Effectively, it involves making land use decisions about an area based on biological, environmental and anthropogenic attributes of the land parcel itself and its surroundings (Desmet *et al.* 2002). Frequently this results in the designation of a specific area of land to be managed for conservation, commonly known as easements, conservancies or concessions, which are one of the main tools for conserving biodiversity on private land (Rissman *et al.* 2007). Easements are used extensively in the United States to ensure that private lands contribute to conservation (Rissman *et al.* 2007). Indeed in the US in 2005, local, state and national land trusts held over 37 million acres for conservation in over 1667 land trusts (Land Trust Alliance 2005). In Namibia, about 80,000km^2 (10% of the country) is under conservancy status (Nuding 2002).

Often nongovernmental organisations (NGOs) or charities are important in leasing or buying land for conservation. In Mexico for example, NGOs have become major leaders in purchasing and managing wildlife habitats and acquiring conservation easements (Valdez *et al.* 2006). The charity Conservation International (amongst others) is pioneering an approach to create a market in biodiversity by leasing land or development rights from landowners for conservation (Ellison 2003). Two examples include a 54,656 acre concession in the Peruvian Amazon, initiated in July 2001 to protect biodiversity, and a 197,600 acre concession in southern Guyana initiated in 2002 to form a key part of the Guianas Tropical Wilderness Corridor (Ellison 2003). Such schemes could potentially be very successful in the Amboseli-Tsavo Ecosystem as well, and the data provided in this thesis would constitute valuable background information necessary for the implementation of such a large-scale project.

Whatever their ultimate objective, conservation plans should be data-driven, target-directed, efficient and flexible (Margules & Pressey 2000), and need to balance biodiversity conservation with other land-use needs. The aim should be to achieve the conservation target with minimum opportunity costs (Faith, Margules & Walker 2001). Conservation planning should also include plans for maintenance of ecosystem services (Singh 2002) and should be based on principles of community based conservation in areas where conservation efforts target community lands (Hackel 1999). Both these issues are discussed below after an examination of the economic principles of conservation.

6.2.1 The economics of conservation

The recent focus in the literature is on the economics of conservation planning (Balmford *et al.* 2000; Moore *et al.* 2004; Naidoo *et al.* 2006; Naidoo & Ricketts 2006; Murdoch *et al.* 2007). Since conservation needs, both at a national and global level, far exceed the available resources, it is vital that scarce resources are used cost-effectively (Naidoo & Ricketts 2006; Murdoch *et al.* 2007). Prioritisation of conservation efforts is therefore crucial and should be based on economic as well as biological information (Balmford *et al.* 2003). Indeed Naidoo & Rickets (2006) define conservation planning as *"the science of systematically prioritizing conservation interventions"*.

Cost efficiency takes into account both the costs and benefits of the proposed action. Murdoch *et al.* (2007) suggest using a return-on-investment approach to prioritize conservation actions. This requires the identification of a well defined and quantifiable objective as well as a realistic estimate of both costs and benefits (Murdoch *et al.* 2007). The financial costs of a conservation activity can be fairly straightforward to calculate (see Naidoo *et al.* 2006), but the benefits are usually a lot more subjective. These should include both the probability of success as well as the 'weight' of importance of the species or area to be conserved. Balmford *et al.* (2003) suggest one measure of conservation benefit is the total area that could be conserved for a given investment.

Margules & Pressey (2000) present a comprehensive review of systematic conservation planning, i.e. how to achieve a given conservation target at least cost. Such techniques are now used routinely in conservation planning (Moore *et al.* 2004; Naidoo & Ricketts 2006) and follow six basic stages; 1) a compilation of data on the biodiversity of the region, 2) an identification of quantifiable conservation goals for the region, 3) a review of existing conservation areas, 4) a selection of additional conservation areas, 5) the implementation of conservation activities and 6) the maintenance of the values of the conservation area (Margules & Pressey 2000). Steps one and two have been achieved in this thesis, as has step three, although it was not documented in detail due to other comprehensive reviews (Croze *et al.* 2006; Kenya Wildlife Service 2007). Step four is described in the following chapter. This thesis has therefore provided much of the background information necessary for implementing effective conservation activities, based on systematic conservation planning. These activities are currently (November 2007) being undertaken on Mbirikani Group Ranch, following the recommendations laid out in Chapter 7. A further discussion of how economic costs have been integrated into the proposed conservation plan for Mbirikani is discussed in the following chapter.

6.2.2 Ecosystem services

Conservation of ecosystem services is another concept which is becoming increasingly important to modern conservation plans (Singh 2002; Kremen 2005; Chan *et al.* 2006; Egoh *et al.* 2007). Ecosystem services are defined as '*natural processes through which ecosystems sustain and fulfil human life*' (Ricketts 2004), and are often resources we take for granted (Ecological Society of America 2000). Many of these are critical to our survival (e.g. climate regulation, air purification and crop pollination) (Kremen 2005), but human influences are so great that in many cases the capacity of ecosystems to provide these critical services is compromised (Daily 1997).

The Millennium Ecosystem Assessment divides ecosystem services into four groups; regulating, provisioning, cultural and supporting (Millennium Ecosystem Assessment 2003), and these are reviewed in Egoh (2007). Regulatory services would include carbon sequestration, pollination, water production, flood prevention, drought prevention and erosion control, while provisioning ecosystem services may include forest production and economically or medicinally useful plants (Ricketts 2004; Egoh *et al.* 2007). Supporting ecosystem services include productive soils and nutrient cycling (Millennium Ecosystem Assessment 2003) while cultural services include aesthetic values, cultural values, ecotourism and recreation (Cowling *et al.* 2003; Egoh *et al.* 2007).

Currently, there is little detailed ecological understanding of most ecosystem services (Kremen 2005; Egoh *et al.* 2007), which impedes progress in their maintenance and management (Luck, Daily & Ehrlich 2003). There is still a need to work out the details of ecosystem services (Singh 2002) such as exactly how vegetation affects soil formation, or exactly which species work best for water retention or carbon sequestration. Nonetheless, ecosystem services are considered important to provide a balanced approach to conservation due to limitations of the biodiversity-centred approach (Singh 2002) which concentrates only on identifying hot-spots of high species diversity. Moreover, the inclusion of ecosystem services into conservation planning would place a specific focus on safeguarding human wellbeing and this might contribute to improving the relevance of conservation to society, leading perhaps to greater support for the conservation intervention (Egoh *et al.* 2007). This may be valuable in areas where local communities see conservation as a threat to their livelihoods and wellbeing.

Although the conservation plan proposed for Mbirikani Group Ranch (Chapter 7) does consider the value of ecosystem services, a much more detailed assessment into ecosystem services in the Amboseli-Tsavo Ecosystem would be very valuable, with a view to incorporating them specifically in future conservation interventions.

6.2.3 Community-based conservation

Once an economic survey and biodiversity study have been undertaken and the need for some kind of conservation intervention agreed upon, the local community, if not already involved, must be included in any further decision making. In a developing country environment, this inclusion of local stakeholders is commonly termed community based conservation, the idea of which was largely prompted by the IUCN World Conservation Strategy published in 1980. This argued a new concept; that successful environmental conservation is reliant upon the active involvement and participation of local communities (McCabe, Perkin & Schofield 1992). Over the past two decades it has become increasingly understood that rural people must play an integral part in conservation efforts (Western & Wright 1994). The old protectionist strategy of fences and fines has no place in a modern society (Adams & Infield 2003) but rather the new community based conservation has gained popularity. Community based conservation (CBC) initiatives work in three ways: they allow people living near protected areas to participate in land use policy and management decisions; they give people proprietorship over wildlife resources; and they ensure that local people receive economic benefits from wildlife conservation (Hackel 1999). CBC should be a bottom-up approach, i.e. decisions regarding land use should not be imposed on communities from above but rather stem from the communities themselves (Western & Wright 1994; Hackel 1999). Indeed decentralisation of resource management from the central authority to local communities is essential for a successful CBC program (Hackel 1999).

The conservation plan for Mbirikani Group Ranch, outlined in the following chapter, makes use of the principles of CBC. The initial idea to set up a conservation area came from a joint meeting of tourist operators, the group ranch committee and the local chiefs, and the large size of the conservation area (70,000 acres) was proposed by the community at a meeting of over 200 representatives. In recognition of the community's proprietorship over their wildlife resource, the land within the conservation area would be leased at fair price, such that the wildlife resource becomes an economic advantage to the local people. All extra revenue generated by this resource would be directed back to the community through employment, education bursaries and provision of infrastructure. In addition, landowners within valuable wildlife corridor areas would be paid to leave land unfenced and uncultivated in recognition of the opportunity cost of doing so.

The latter point effectively constitutes payments for conservation friendly practices, and is a method used throughout the world, in terrestrial and marine ecosystems. An example is the Kitengela Lease Program in Kenya, where voluntary participants who live in the dispersal area for wildlife from Nairobi National Park are paid $4 per acre per year not to fence, develop or sell their land, although they may continue to live there and graze their livestock (Ole Nkedianye 2003). In this way, the community ownership of the wildlife resource is recognised, and financial benefits offered in exchange for conservation-friendly practices. The idea evolved initially from the community. A similar situation is found in the European fishing industry, where new reforms are promoting financial incentives to restrain fishing efforts and guide the industry towards more responsible and sustainable fishing practices (Symes & Pope 2000).

Community based conservation is thus an optimal template to follow, applicable not only to Maasai areas, or Kenyan savannahs, but throughout Africa and the rest of the world where there is a need for conservation of resources that are depended on by people for their survival. Indeed this model has been applied successfully in many parts of the world. One of the most famous examples is the CAMPFIRE initiative in Zimbabwe (Child 1995) where the allocation of user rights is to local landholders, and economic benefits at the household level are maximised to generate the incentives to conserve the resource (Child 1996). In Mexico, the new wildlife conservation initiative, 'Wildlife Conservation and Production Diversification in the Rural Sector', promotes participatory conservation opportunities by involving key stakeholders in management decision (Valdez *et al.* 2006). In Namibia, where 75% of the wildlife is found outside formally protected areas (Nuding 2002) and the local farmers suffer the costs, it was recognised that, as well as increasing economic benefits to local people, rural communities must be given a role in the management of the wild animals that share their land (Nuding 2002). Namibia's solution to this is wildlife conservancies whereby neighbouring landowners or members of communal land combine their natural resources for the purposes of *"conserving and using wildlife on a sustainable basis"* (Nuding 2002).

6.2.4 Summary

Effectively, conservation planning should be based on sound economic principles and include provisions for the maintenance of ecosystem services as well as biodiversity. The principles of community-based conservation should be incorporated where local landowners are likely to be affected by a conservation intervention. An example of how this can be achieved is presented in the following chapter.

6.3 SUGGESTIONS FOR FURTHER RESEARCH

For many of the areas where further research was considered necessary, it has been mentioned in the relevant section in the discussion above. The following is a summary of these, and some additional recommendations for future research, which have arisen from this thesis.

Future conservation work within the Amboseli-Tsavo Ecosystem would benefit from an inventory of the ecosystem services present including their level of threat status. For example it would be beneficial to quantify the value of the different habitats for carbon sequestration or water retention, or the different plant species for their contribution to erosion control, pollinator survival or their medicinal or economic value. Likewise, conservation economics could be applied with greater rigour than in the current study. It would be very worthwhile to design and implement a study to investigate the costs to Maasai livestock farmers of competition for grazing with wildlife.

Importantly, further research should focus on the exact design, location and management of corridor areas to maintain the link between the Mbirikani Conservation Area, Amboseli and Tsavo. Such research may benefit from the use of GPS collars on the main migratory species, zebra and wildebeest, to investigate exactly how they move around the ecosystem, and the extent to which they are displaced by human influences. Likewise the use of GPS collars to investigate the movement patterns of the vulnerable fringe-eared oryx (see Groom, Hill & Bonham 2007) would be valuable.

Finally, a better planned study and more detailed analysis into the effects of subdivision on the distribution patterns of wild grazers would be very useful. It would be interesting to investigate this for ranches of different productivity. Additionally, it would be interesting to investigate the effects of land subdivision and sedentarisation on the quality of the grass resource.

6.4 CONCLUSION

The region of East Africa which spans the Kenya-Tanzania border and includes the ranches studied in this thesis has the richest mammalian fauna on the planet (Little 1996; Boitani *et. al.* 1998). Conservation of this ecosystem is therefore of major international importance. I have illustrated in this thesis the conservation threat posed to the area by land subdivision and have highlighted the need for management of this process. Moreover I have attempted to provide sufficient data to enable sound, practical and effective conservation work to be implemented.

Localized approaches are always essential when developing conservation plans for specific areas, and for this reason Chapter 7 goes into details of how conservation could best be implemented on Mbirikani. However, this chapter and Chapter 1 both discussed the global relevance of the issues and conservation threats tackled in this thesis, and I believe there are some general lessons from the thesis which could be applied throughout the world, in similar situations. These include the necessity to use a holistic approach, including ecological, economic and social considerations. Whilst a detailed understanding of the ecology, ecosystem processes and ecosystem services is a vital prerequisite for any conservation plan, an investigation of the underlying economic context is also essential. The value of investigating local attitudes and opinions, and more importantly utilising expert local knowledge at all stages of the design of conservation interventions cannot be underestimated. From a practical point of view, in order to increase tolerance of wildlife within local communities, increasing the spread of benefits and trying to ensure an equitable distribution could have a considerable impact on people's attitudes for very little extra investment.

The discipline of conservation planning has recently taken on a more rigorous and structured approach and following the three main principles discussed in this thesis is now recommended. The reviews of conservation economics, preservation of ecosystem services and community based conservation should be of use to anyone attempting conservation work at almost any scale. For those attempting small scale initiatives on communally owned rangelands, the following chapter provides a useful case study.

CHAPTER 7

A CONSERVATION PLAN FOR MBIRIKANI GROUP RANCH

7.1 INTRODUCTION

One of the major aims of this thesis was to produce a conservation-based management plan for Mbirikani Group Ranch as it undergoes land subdivision. This chapter presents one potential conservation plan, based on research findings and experience, and incorporating the conservation principles discussed in the previous chapter. The broad principles of the plan are applicable throughout the rest of the Amboseli-Tsavo Ecosystem.

The chapter will focus exclusively on Mbirikani (and not Merueshi) for two main reasons. Firstly, when the research project was initiated, it was with the overall aim of helping to secure the future of wildlife on Mbirikani Group Ranch at the time of land privatisation. Merueshi was used as a case study to investigate what can happen if subdivision occurs without any conservation management strategy in place. Secondly, as a conservationist, it is far more worthwhile to focus on Mbirikani than on Merueshi, for the reasons below.

Findings presented in Chapter 2 show that Merueshi supported only half the density of wildlife that Mbirikani did. Merueshi was also very fragmented, with almost 100km of fencing in place. Additionally, on Merueshi Group Ranch, community support for wildlife conservation was very low (see Chapter 5, Figure 5.2). Although 55% of people agreed that wildlife could bring a lot of income to their households if it were well managed, only 14% agreed that wildlife should be conserved on Merueshi, with only 14% also agreeing a specifically designated conservation area would be a good way of doing this. With such little support even for the idea of wildlife conservation, the likelihood of success of a conservation project in the area would be much lower than in other places. It may be that if it could be done gradually, people would change their minds as they started to receive income from wildlife but, with limited funds available, it makes greater economic sense to employ these where a) there is more wildlife and b) community support for conservation is higher. Moreover, the spatial location of Mbirikani makes it potentially a more useful site for conservation due to its proximity to the Chyulu Hills and Tsavo West National Parks. In this region therefore, Mbirikani is a far more suitable candidate for conservation than Merueshi, although ideally if the program is a success for Mbirikani, conservation efforts should expand to the surrounding ranches, including Merueshi.

7.2 INTEGRATING ECONOMIC COSTS INTO CONSERVATION PLANNING FOR MBIRIKANI

Chapter 6 discussed the importance of integrating economic costs into conservation planning. The major relevance of this however, is at a much greater scale than this project, i.e. when deciding which regions within which countries should be prioritised for conservation. For example, the principles of Murdoch *et al.* (2007), of using a return-on-investment approach are applicable mainly at a much broader scale (i.e. nationally or globally), rather than locally, where there are many other factors which need to be taken into account. Indeed Naidoo *et al.* (2006) state that the importance of including costs in conservation planning depends on the spatial correlation between biological benefits and costs, something that is likely to vary less over a small spatial scale than over a larger one.

The decision to try and conserve areas within the Amboseli-Tsavo Ecosystem for wildlife was taken long before this project was initiated (R. Bonham *pers. comm.*). The goal of this project was to investigate the most effective way to go about doing it. Nonetheless, the conservation plan for Mbirikani (presented in the following section) does use economic principles. Costs of conservation include land prices, management costs and opportunity costs (Naidoo & Ricketts 2006), while benefits include the quantity and diversity of biodiversity protected, maintenance of ecosystem services and the amount of land protected. On Mbirikani, the location of the proposed conservation area within the ranch incorporates the greatest number of different habitats, the highest concentrations of wildlife, the greatest diversity of wildlife and the most biologically under-represented habitats in the ecosystem (the lava forests). It is likewise in an area where land prices are lowest (because the lava forests are virtually worthless for livestock grazing and there is no permanent water), opportunity costs are fairly low (very few households have permanent bomas in the area), and management costs would be lower than elsewhere on the ranch because the infrastructure is already in place (from the current conservation efforts of the single safari lodge). Therefore the location of the proposed conservation area (see Figure 7.1) is optimal from both a biological and an economic perspective.

However, even if this were not the most economically beneficial area in which to target conservation efforts, it is the *only* location which would be allowed by the community, because the rest of the ranch is too heavily utilized by the people and their livestock and they would not tolerate having such land removed from their use for conservation.

Additionally, it might make more economic sense, and even conserve land of greater biological potential, to lease or buy the land around Amboseli National Park for conservation. However, even if the return on investment were considerably greater than other options, this would not currently be a feasible solution without alienating the local people, who already feel their rightfully owned land has been removed for conservation (Western 1994). Taking into account costs of lost community support and/or benefits of improved community attitudes towards conservation is very important in areas where conservation activities need to work very much alongside the needs of the local people. As such, when dealing with local communities, one's choice of focus for conservation efforts cannot solely be dictated by economic principles.

7.3 CONSERVATION ON MBIRIKANI GROUP RANCH

Land privatisation is inevitable and imminent on Mbirikani Group Ranch, and the need for a conservation plan has been made clear. The challenge is therefore to create a realistic, feasible and economically sound conservation plan for Mbirikani during and post land privatisation.

The first step in implementing any such conservation plan would be to assess the attitudes and opinions of the local stakeholders (group ranch residents), because no conservation activity in this situation would be successful without community support. In many parts of Africa, for example, wildlife conservation has lost favour with local people because it has put the needs of wildlife above those of people (Abrahamson 1983; Hackel 1990). However, results presented in Chapter 5 (Figure 5.2) show that the vast majority (84%) of Mbirikani household heads believed that wildlife could bring a lot of income to their household if it were properly managed. Furthermore, over 70% wanted to conserve wildlife on their ranch after land subdivision and 54% were in favour of setting aside a specific conservation area to do so. These very positive findings suggest that a properly designed and managed conservation program on Mbirikani would have a good chance of success.

An analysis of the ecology of the landscape involved, the economic situation, people's attitudes towards wildlife and conservation, and the results of countless informal conversations with the ranch members and their leaders, suggest the following broad conservation strategy: a wildlife conservation area should be demarcated within the ranch from which every member can benefit. The remainder of the ranch would then be privatised, with wildlife corridors maintained to link the area with Amboseli and Tsavo National Parks. With this in mind, the remainder of this section uses results from the thesis to suggest the optimal design for 'The Mbirikani Conservation Area' followed by suggestions for conservation on the rest of the ranch. First however, I briefly describe the background to conservation efforts on Mbirikani Group Ranch, to put the current situation into context.

7.3.1 Background to conservation efforts on Mbirikani

I have just mentioned the considerable community support for conservation on Mbirikani. This stems from a long history of conservation efforts on the ranch (21 years), the successes and failures of which have led to the current situation and suggestions for the future.

Wildlife has been utilised as a resource on Mbirikani Group Ranch since 1986 when Ol Donyo Wuas Lodge was first built. From this, the community (via the group ranch leadership) received rental fees for the land and bed night fees from the guests. However, this was on a fairly small scale (only 8 beds). The wildlife revenue generated on the ranch increased in 1991 when the Maasailand Preservation Trust (now Ol Donyo Wuas Trust) was initiated. At first this included rewards for snares collected, the creation of a women's group and a workshop for the women's craft activities. The Ol Donyo Wuas Lodge expanded to its present size (20 beds) in 2005, and sponsorship of students started in 1996, with five students. Revenues increased again in 1997 when nine local game scouts were employed on Mbirikani. Since then, the Trust has been continually expanding, especially through the provision of educational scholarships and an expanding team of game scouts (now 56). Conservation benefits took a big leap forward in April 2003 with the initiation of the Predator Compensation Fund, which has taken the Trust to another level. Not only has it paid out around $30-40,000 per year since its initiation (Hill 2006), the fact that it is available to everyone on the ranch is of immense importance. No longer were conservation benefits restricted to a minority of people fortunate to secure jobs or education bursaries, but the infrastructure was there for *everyone* to receive payments in compensation for livestock depredation by carnivores.

Having a legally demarcated conservation area on Mbirikani would take this one step further. Revenue from wildlife would actually be received by every household, constituting a regular and reliable source of income, irrespective of environmental or political conditions. Additionally, the payments would be equally distributed amongst households, which is envisaged by many to be a means of reducing the inequality between rich and poor. Since the wealthy elites ('cattle barons') constitute only a small minority of the overall population, a situation like this would be favoured by the vast majority of members.

7.4 THE MBIRIKANI CONSERVATION AREA

7.4.1 Selecting the location

Mbirikani Group Ranch is approximately 320,000 acres in size, with approximately 4,650 members in 933 households. Already, the 4,000 acres or so of irrigable land has been allocated amongst the members, leaving 316,000 acres of communal land. This would equate to each member receiving around 68 acres of land, were the land allocation to take place on the basis of ranch size divided by number of members.

Instead, it has been decided (by the committee and elders representing their local communities) that there should be a conservation area on the ranch. Results presented in Chapter 3 (Figure 3.1) illustrate clearly the great importance of part of the eastern edge of the ranch for wildlife. This is also where the current safari lodge is located, and where there is very little permanent human settlement (see Chapter 2, Figure 2.5). Moreover, this eastern section of the ranch lies adjacent to the Chyulu Hills National Park, which has its own unique flora and fauna, and would effectively considerably increase the size of the protected area. It is therefore proposed that this is the optimal location for a wildlife conservation area and indeed this is the area proposed by the community as well.

Loss of heterogeneity can be a problem when selecting relatively small conservation areas, but the area proposed is actually one of great habitat diversity and would include between four and six of the eleven different habitat types on the ranch, depending on the size. Arguably, with the possible exception of the riverine area in the south western corner of the ranch, these are the most valuable habitat types in terms of grass quality and quantity, as well as the areas most utilized by wildlife (see Chapter 3, Table 3.2).

Preservation of ecosystem services must also be considered when choosing the location for a conservation area. Different types of ecosystem services were reviewed in Chapter

6. The regulatory services of carbon sequestration and water production would be best maintained by conserving this area at the base of the Chyulu Hills, because this serves as the ecosystem's main watershed and includes the most densely vegetated habitats (lava forests), which would be most valuable for carbon storage. Likewise the array of different habitats to be protected by this choice of location would ensure the greatest protection for provisioning ecosystem services such as medical or economically important plants. Cultural ecosystem services would also be best maintained by this choice of location due to the immense beauty of the area and its value for tourism.

An area along the eastern boundary of the ranch is therefore selected as the optimal location for the conservation area; a decision agreed upon by conservationists, researchers, tourism operators and the Maasai community. This is illustrated in Figure 7.1.

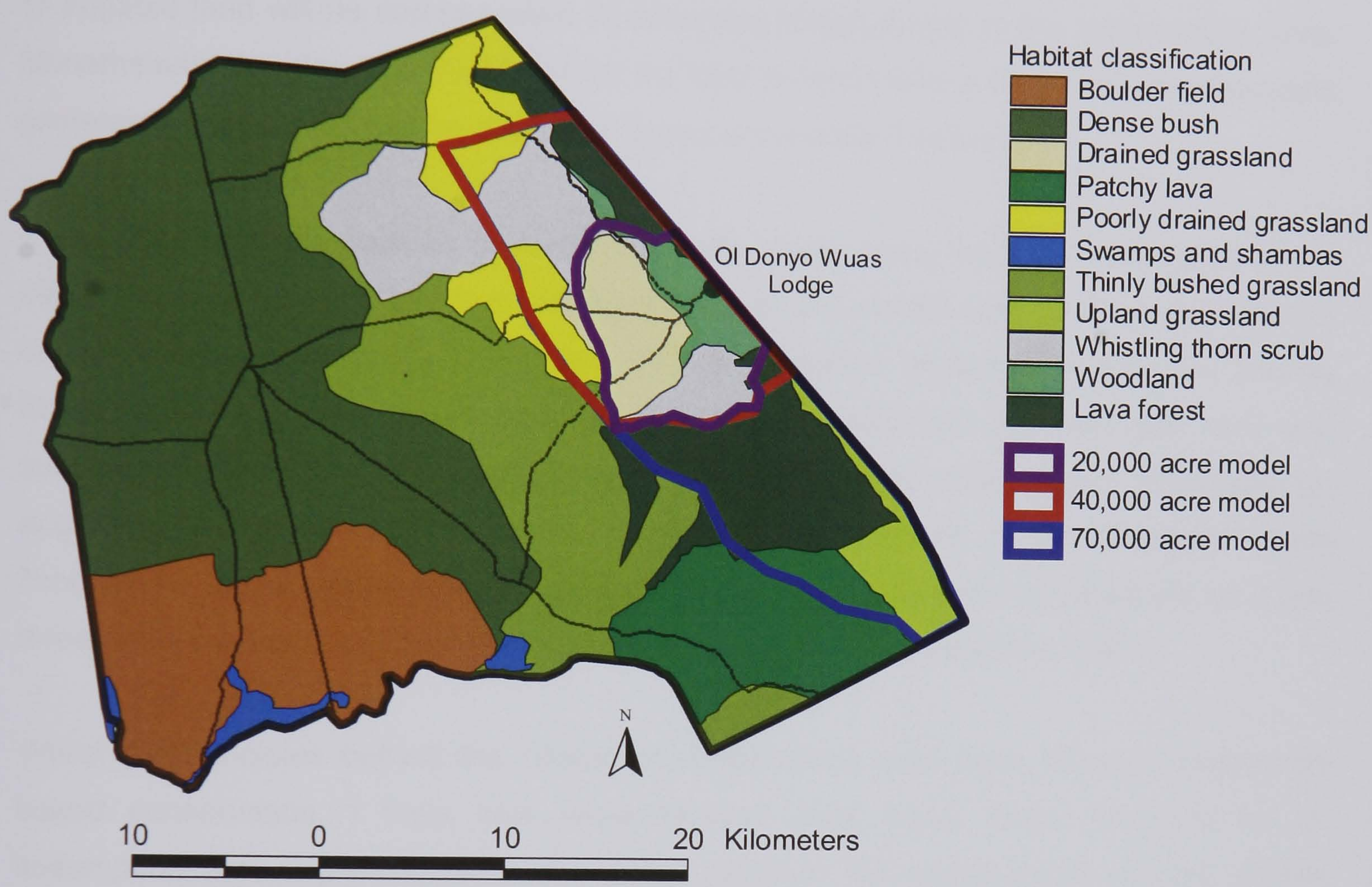
7.4.2 Selecting the size of the conservation area

There are several options for the size of the conservation area; 20,000, 40,000 and 70,000 acres. Tomlinson, Hearne & Alexander (2002) suggest that small wildlife reserves of 7410 acres or less may not be justified on the basis of profits alone. Although the proposed Mbirikani Conservation Area is about more than profit, nonetheless the minimum size area being suggested is 20,000 acres.

If a 20,000 acre conservation area was decided upon, this would entail each member giving up 4 to 5 acres of their allotted 68 acres to the conservation area. This is an average figure, however, because the unequal distribution of irrigable plots means some people are 'owed' more than others in the conservation area. Irrespective of the exact division of shares however, the idea would be to have the entire area as a 'no boma, no grazing' zone, i.e. to have exclusive use by wildlife.

Two other models are also being proposed. One is a 40,000 acre conservation area, which would be an extension of the 20,000 acre model (see Figure 6.1). The other, proposed by the Maasai community themselves, is a far larger area extending the entire way down the base of the Chyulu Hills to the neighbouring group ranch in the south. This would cover an area of approximately 70,000 acres, and entail each member giving up on average 15 acres of their allocated land. However, this might be a very good option for all concerned, because about 24,000 acres of that land would be the lava forests. These acres have virtually no grass and therefore represent only a small loss of grazing resources to the Maasai. However, they are botanically rich (see Appendix 2B) and provide a very important refuge for carnivores, especially lions, as well as buffalo, elephant and klipspringer.

Figure 7.1 An illustration of the three different conservation area models for Mbirikani Group Ranch



7.4.3 The legal infrastructure of the Mbirikani Conservation Area

Whichever option is eventually chosen, I advocate certain principles for how the conservation area is legally structured and utilized, which is applicable to all models. Firstly, the conservation area should be legally leased from the Maasai on a renewable lease of less than 10 years. This is to avoid the higher tax duty payable on leases over 10 years in length (Kenya Land Office, *pers. comm.*), and also to ensure that the plan is not seen as a way of taking land from the Maasai, as long leases are often considered to do.

Ideally, each member should hold an equal share in the conservation area, i.e. an equally sized plot of land. However, the land holdings within the conservation area should be unspecified with regard to exact location, such that individuals cannot lay claim to better or worse places and thus demand different lease payments or feel unfairly treated. The conservation area should work as a company, being registered as a land trust, with each member of Mbirikani Group Ranch holding one share (e.g. see Mwathi *et al.* 2005). Unfortunately, in reality, during the first phase of land subdivision in 2005-6, which was to subdivide all irrigable areas on the ranch, plots were not allocated equally amongst the members. It has therefore been communally agreed that those who received smaller plots of irrigable land will be compensated by receiving larger shares in the conservation area. Nonetheless, the idea of a trust in which the land is held under a single title deed remains preferable, and shares can be allocated disproportionately if necessary.

Under this scenario, instead of receiving a title deed, every member receives a share certificate in which their right to a holding equivalent to a certain number of acres within the conservation area is legally presented. The conservation organisations that are funding this conservation area would need to guarantee a minimum payment per acre and subsequent fees generated would also be split between all shareholders proportionally, according to the size of their share. Following the example of the Kitengela Lease Program for the dispersal area of Nairobi National Park, payments would ideally be made three times per year, just prior to each new school term (Ole Nkedianye 2003).

Whilst the principles behind the choice of conservation area were those of community based conservation, I have also recommended legal input. There tends to be an assumption that implementation of a CBC program will automatically ensure wildlife protection (Hackel 1999), but I believe this is not always the case. Whilst CBC is clearly extremely important, it has been suggested that it is being oversold, and the need for protectionism understated (Hackel 1999). My suggestion for the conservation of Mbirikani is to adopt both approaches: the community should decide on the location and size of their conservation area, which would then be protected under a set of rules and regulations decided upon by the community and the conservation organisation who would be leasing it. After this, the agreement should be made binding by law, such that the protection afforded to the conservation area can be legally enforced. This is very important, as it would ensure that the fickle nature of local politics (i.e. changes in local leadership, tribal conflicts etc.) cannot affect the conservation decision previously agreed on by the entire community. Of course this latter point is critical; it is vital to ensure that any such decision is indeed agreed upon by the majority of the community members, and not dominated by the political and social elites as so often happens.

7.4.3.1 Community use of the conservation area

The issue of cattle grazing within the conservation area also needs to be legally outlined. Should the conservation area only be 20,000 acres, it would be best to maintain a strict no-grazing policy within the entire area. However, in either of the two bigger scenarios, there should be a grazing policy in place which outlines the use of specific areas for cattle at certain times of year. This may be flexible if it is dependent on resource availability outside the conservation area, such that during drought years, more cattle are allowed into the conservation area. However, following the example of Koyiaki Conservancy, each head of cattle grazing inside the reserve would be charged around 6-8 Kenyan shillings (US$0.1) per day as a grazing fee (Ron Beaton, *pers. comm.*). This money would then add to the revenue earned by the conservation area, being distributed to the individual members at the next payout day. This helps to equalise, to a degree, the inequality on the ranch between those rich members with large herds of cattle and those with only a few.

In addition, grazing by cattle at an intermediate level based on a rotation system should maintain a high quality of grass in the conservation area (Guevara *et al.* 1996). In the core of the conservation areas, where no cattle are allowed, it may be necessary to carry out periodic controlled burning to manage the grass if wildlife grazing is insufficient. Another alternative, possibly the preferable one, would be to let cattle into these areas during the wet months when the lodge is closed to tourists. The grazing fees generated would increase the total revenue returned to the community from the conservation area, the quality of the grass would be maintained, and the community would benefit by having this extra grazing resource. Such an open system might also encourage the feeling of a joint venture; not simply more land being removed for conservation. In any event, the use of the conservation area by cattle would have to be heavily regulated and efficiently enforced by a team of game scouts.

7.4.4 The economics of the plan

In 2005, households on Mbirikani Group Ranch faced a mean annual deficit cost to wildlife of almost $400 per year (Chapter 4, Table 4.7). It is possible that the demarcation of a specific conservation area would reduce these costs through limiting the contact between people and wildlife. However, the conservation area would not be fenced and undoubtedly some wildlife would persist on the remainder of the ranch, so households would still incur costs from wildlife. One of the major aims of the conservation area must be to pay households enough to make it worth their while for a) sacrificing grazing land for conservation and b) continuing to incur costs from wildlife.

Potential investors are offering to pay around \$5 per acre per year for land within a defined conservation area. For the model with the 20,000 acre conservation area, this would mean a total of \$100,000, which would equal just over \$100 per household per year. If 40,000 acres were to be demarcated as a conservation area, at the same price, a total of \$200,000 would be generated and households would receive around \$200 each per year. If the 70,000 acre model was chosen, total lease fees would be \$350,000 and households would receive about \$375 every year. Although the latter scenario comes close to balancing out the *mean* costs from wildlife (from disease and predation), for some households (those that incur the highest costs) it would be far from sufficient. For many households however, it would constitute a substantial profit. In either scenario, it would still be a major improvement on the current situation, and results from Chapter 5 suggest that this would be sufficient to further improve attitudes towards wildlife and its conservation, especially since it would be equally distributed at the household level.

The lease payments would be funded mostly by tourist bed-night fees. Ol Donyo Wuas Lodge is a 20 bed lodge. Assuming an average of 50% capacity throughout the year, with clients paying a \$60 conservation fee per night, this would generate \$219,000. From the figures above, it is clear that this would be sufficient for the 20,000 or 40,000 acre conservation area models, but were the 70,000 acre model to be chosen, more money would be needed. This would probably have to come from increasing the capacity of beds, either by expanding Ol Donyo Wuas Lodge or by bringing in a second investor to set up another lodge in the conservation area. Alternatively other sources of funding would need to be secured.

In addition to the lease payments, the Ol Donyo Wuas Trust would continue to operate. In 2005, this Trust generated almost \$100,000 for the community through predator compensation, education bursaries, game scout wages and teachers salaries (from Chapter 5, Table 5.1). Adding this to the lease fees would mean that for the vast majority of households, wildlife-generated revenues would exceed costs. For the larger herd owners in high conflict areas however, this would still be insufficient, especially considering the opportunity cost from losing grazing land to wildlife.

7.4.5 Summary

Currently, the larger conservation area (of about 70,000 acres) is the one being favoured by the community. The very fact that people are trying to secure as much land as possible in the conservation area (at the expense of getting land elsewhere) suggests that they predict not only the success of the venture, but the generation of considerable revenues from wildlife. This may be partly because people have recently learned the extent of revenue being generated by wildlife ($230,450 in 2005 - see Chapter 5), and understand that, with individual ownership of land or individual shareholdings in a conservation trust, some of these benefits would become available to them directly.

7.5 CONSERVATION IN THE REST OF THE RANCH

Even with a formally designated and protected conservation area, the overall goal of ecosystem level conservation requires a broader strategy. I envisage three further goals: to keep the remainder of Mbirikani's rangelands as open as possible, to secure access to water for wildlife and to maintain wildlife corridors. Keeping the rangelands open does not mean opposing land privatisation, but rather opposing the fencing of land parcels and encouraging the continuation of ranch-level rotational grazing systems and reciprocal use of neighbouring land to avoid the problems of consistent pressure on the land. Chapter 2 describes the negative ecological consequences of fencing and reduced mobility of livestock.

Unless artificial water points are constructed in the conservation area, the only access to water for wildlife in the dry seasons would be from the Kikarangot River in the south of the ranch, which is outside any of the proposed conservation area models. Consequently access to water for wildlife would need to be secured independently, possibly by leasing a one kilometre stretch of land adjacent to the river, to keep it open for wildlife. However, without maintaining the connectivity of Mbirikani with the rest of the ecosystem such conservation efforts will have minimal success. The following section discusses the importance of wildlife corridors as a conservation tool in the area.

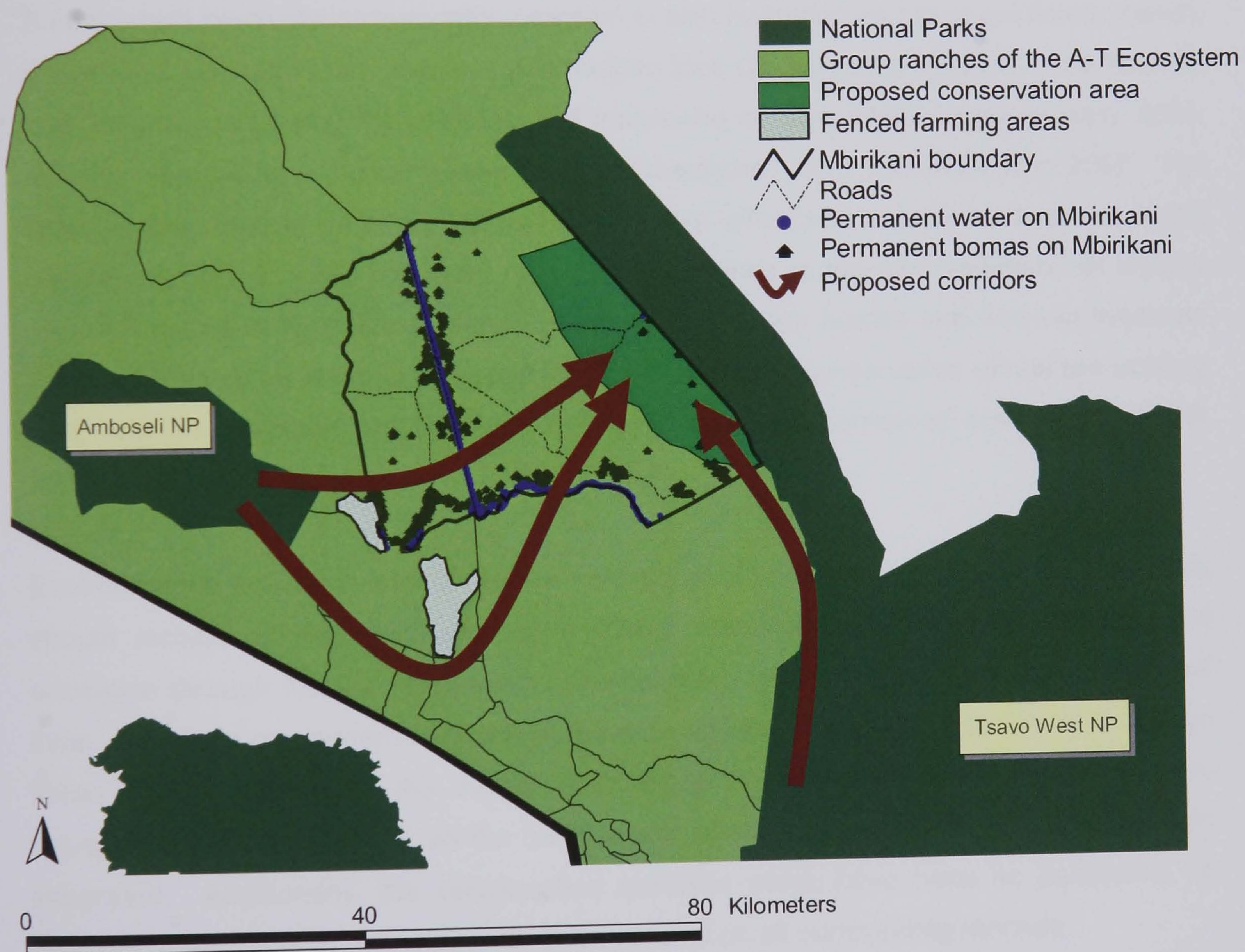
7.5.1 Wildlife Corridors

Over 70% of the total wildlife population on Mbirikani in the wet season consisted of zebra and wildebeest. Whilst approximately half of these were resident individuals, also present in the dry season, 50% (over 10,000 individuals) moved into the area at the beginning of the rains. The corresponding decrease in zebra and wildebeest populations in Amboseli (D. Western, *pers. comm.*) and the long lines of animals walking east from the park suggest that many of these animals came to Mbirikani from Amboseli, although it was known that some also came up from the south, from Tsavo. There was also a migration of elephants from Tsavo, and the fringe-eared oryx and eland both moved out of the study area twice a year for about two months at a time, although it is not clear where they went to. Further research into the movement patterns of these species would be very valuable because if they are to be protected in the future, their movement patterns need to be understood.

The importance of maintaining links with the rest of the ecosystem is therefore clear, and use of wildlife corridors would be an appropriate tool. Exactly where the corridors should be located is a matter for further research, although there are limited areas left where wildlife is still able to move through the dense human settlements along the pipeline road and the river, and every effort should be made to keep these areas open. Suggested corridor areas are illustrated in Figure 7.2.

Whilst land privatisation can potentially cause land fragmentation and ecological degradation (Chapter 2), for the purpose of implementing the three conservation strategies outlined above, private land ownership could actually be advantageous. The main advantage of this system is that conservation managers can deal with individual land owners who have the right to make independent decisions about their land, rather than trying to deal with a (frequently corrupt) committee. For example, landowners in key areas could be paid a small fee to prevent fencing or cultivation which would disrupt wildlife movements. Such intervention may be considered payment for ecosystem services, by allowing a conservation manager to compensate a landowner for preserving a certain service, in this case the grazing resource and openness of the landscape. This is the underlying premise of the Kitengela Lease Program near Nairobi, where landowners are paid 300 Kenyan shillings ($4) per acre per year to leave their land unfenced (Ole Nkedianye 2003). For Mbirikani, it would be ideal if this money could come from the revenue sharing program of Amboseli National Park, which seems likely to increase sizably in the near future. Since Mbirikani Group Ranch is one of the main dispersal areas for Amboseli wildlife, it is probable that ranch members would be entitled to a large share of this, but only if they can show serious commitment to the conservation of the wildlife into the future. This is further incentive to carefully plan and manage for wildlife, including keeping corridors open between Amboseli and the proposed conservation area. Alternative sources of funds for leasing corridor land include organisations such as the World Land Trust, which specialise in leasing land for conservation (World Land Trust 2007).

Figure 7.2 Map showing the proposed wildlife corridors between Amboseli and Tsavo National Parks and The Mbirikani Conservation Area.



7.5.2 Other conservation activities on Mbirikani

As demonstrated in Section 7.2.1, conservation has a 21 year history on Mbirikani, with wildlife revenues continuously increasing. Currently, major conservation efforts include payment of teachers' salaries, education bursaries, employment of local game scouts, conservation-oriented research projects and predator compensation. All of these activities should be continued concurrently with the operation of the conservation area.

For example, the predator compensation project had been operational on Mbirikani Group Ranch since 2003 and seemed to be having a positive effect on carnivore conservation. Stopping this program may engender negative attitudes towards the carnivore population and, for this reason as well as its apparent success in increasing tolerance of carnivores, a priority must be to secure long term funding for this program. This would ideally be through the implementation of an endowment fund.

A concurrent carnivore conservation program is also operating on Mbirikani Group Ranch, under the Living with Lions initiative (Kilimanjaro Lion Conservation Project). This has two components, one being the collaring and monitoring of lions which began in early 2004, and the other being a project known as 'Lion Guardians' which was initiated in 2007. The latter is described in detail by Hazzah (2007), but effectively employs a team of young Maasai warriors to track lions and report their locations to the lion biologists, as well as preventing conflict by helping pastoralists to build stronger bomas and find lost livestock. All indications so far suggest this combination of carnivore conservation efforts are working well, and lion numbers on Mbirikani Group Ranch have increased since 2003 (Frank 2007).

Environmental education should also be a priority in any long-term conservation plan. This should include primary and secondary school education programs, as well as adult education through the use of meetings, church gatherings and the showing of educational films. All these components should continue to operate under the umbrella of Ol Donyo Wuas Trust. Indeed for the overall goal of the conservation of the Amboseli-Tsavo Ecosystem to be successful, all the components of the conservation model must be fully integrated. Additionally, the conservation activities which have been so successful on Mbirikani Group Ranch would need to be replicated on all surrounding ranches.

7.6 CONCLUSION

This chapter has used the results of this thesis, knowledge gleaned from the literature and years of experience living in the area to produce a conservation plan for Mbirikani Group

Ranch. Whilst the thesis has been in the final stages of preparation, this plan (Groom *et al.* 2007) has been actively used and implemented by fundraisers, donors and planners alike.

The plan is based on the principles of community based conservation, incorporating conservation economics where appropriate. Taking into account the community's opinions, combined with a background understanding of the distribution of ecological resources within the landscape and the economic context in which conservation would be carried out, the plan outlined in this chapter has the highest potential for long term success.

The plan is also virtually directly transferable to surrounding ranches, and for this reason, with the right investment and management, I believe the outlook for the future of wildlife within the Amboseli-Tsavo Ecosystem is positive. Moreover, lessons from this area could be applied throughout the world's rangelands where land subdivision and / or pastoral sedentarisation are occurring in conservation hotspots.

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APPENDICES

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APPENDIX 2A

DETAILS OF HABITAT TYPES

Mbirikani Group Ranch

Table 2A.1 Vegetation composition of different habitat types on Mbirikani Group Ranch
DB=dense bush, TBG=thinly bushed grassland, BF=boulder field, LF=lava forest, WT=whistling thorn scrub,

	Dominant tree species	Dominant shrub species	No of plant sp	Dominant grass species	No of grass sp
MBI					
DB (444 km^2)	*Commiphora rostrata* *Acacia mellifera* *Acacia nubica*	*Solanum incanum* *Tephrosia villosa* *Barleria acanthoides*	46	*Sporobolus fimbriatus* *Pennisetum mezianum* *Eragrostis keniensis*	26
TBG (226 km^2)	*Commiphora africana* *Acacia mellifera* *Commiphora rostrata*	*Solanum incanum* *Melhania ovata* *Barleria acanthoides*	41	*Pennisetum mezianum* *Bothriochloa glabra* *Cenchrus ciliaris*	36
BF (171 km^2)	*Acacia mellifera* *Balanites aegyptiaca* *Acacia Senegal*	*Cyathula erinacea* *Solanum incanum* *Sansevieria suffruticosa*	48	*Sporobolus fimbriatus* *Dactyloctenium aegyptium* *Setaria verticillata*	13
LF (120 km^2)	*Acacia brevispica* *Pappea capensis* *Dalbergia vaccinifolia*	*Justicia* species *Cissus quadrangularis* Unknown species	61	*Brachiaria dictyoneura* *Sporobolus festivus* *Brachiaria serrifolia*	10
WT (97 km^2)	*Acacia drepanolobium* *Acacia nilotica* *Balanites aegyptiaca*	*Indigofera* species *Cyathula erinacea* *Solanum incanum*	27	*Sporobolus pellucidus* *Pennisetum mezianum* *Chrysopogon aucheri*	28
PL (97 km^2)	*Acacia mellifera* *Cordia ovalis* *Acacia brevispica*	*Tephrosia villosa* *Indigofera* species *Tephrosia* species	49	*Cynodon plectostachyus* *Cyperus obtusiflorus* *Digitaria milanjiana*	23
PDG (70 km^2)	none	*Triumfetta flavescens* *Solanum incanum* *Cyathula erinacea*	16	*Pennisetum mezianum* *Chrysopogon aucheri* *Sporobolus fimbriatus*	20
DG (40 km^2)	none	*Indigofera* species *Hermannia alhiensis* *Solanum incanum*	7	*Sporobolus pellucidus* *Digitaria milanjiana* *Sporobolus fimbriatus*	20
UG (32 km^2)	*Erythrina abyssinica* *Acacia hockii*	*Diplolophium africanum* *Artemisia afra* *Eriosema flemingioides*	22	*Themeda triandra* *Hyparrhenia hirta* *Hyperthelia dissoluta*	15
WL (24 km^2)	*Acacia tortillis* *Acacia mellifera* *Commiphora africana*	*Indigofera* species *Solanum incanum* *Pavonia urens*	40	*Cynodon plectostachyus* *Eragrostis superba* *Microchloa kunthii*	21

PL=patchy lava, PDG=poorly drained grassland, DG=drained grassland, UG=upland grassland, WL=woodland

Merueshi Group Ranch

Table 2A.2 Vegetation composition of different habitat types on Merueshi Group Ranch
DB=dense bush, TBG=thinly bushed grassland

	Dominant tree species	**Dominant shrub species**	**No of plant sp**	**Dominant grass species**	**No of grass sp**
MBI					
DB (70 km^2)	*Commiphora rostrata* *Cordia gharaf* *Lannea flocossa*	*Astripomoea hysocamoides* *Drimia indica* *Ocimum americanum*	43	*Sporobolus fimbriatus* *Eragrostis keniensis* *Microchloa kunthii*	26
TBG (113 km^2)	*Acacia drepanolobium* *Acacia senegal* *Acacia mellifera*	*Astripomoea hysocamoides* *Solanum incanum* *Tephrosia villosa*	37	*Pennisetum mezianum* *Cenchrus ciliaris* *Sporobolus fimbriatus*	30

APPENDIX 2B
COMPARISON OF BELT AND POINT TRANSECTS

Comparison of belt and point transects

Two different methods were employed to count wildlife and livestock each month, belt transects and point transects. These both involved a total count of all animals within a fixed area and consequently should have provided density estimates that were directly comparable. However, it is possible that methodological differences could have led to systematic biases. For example, belt transects necessitate that the observer was moving, potentially increasing the likelihood that individual animals moved away before they could be counted, i.e. belt transects may have under-estimated animal density relative to point transects. In order to determine whether there was any evidence for such biases, density estimates from the two different techniques were compared.

In March, October and November 2005, a series of independent belt and point transects were conducted in two different habitats, open grassland and thinly bushed grassland. As explained in Chapter 2, the data were not normally distributed and therefore had to be analysed using a series of non-parametric (Mann-Whitney) tests (72 in total). Although this involved a multiple testing procedure, the significance level was retained as $\alpha = 0.05$ to minimize Type II errors. The results are shown in Table 2A.1.

These results show that overall there were no significant differences in the density estimates derived from point and belt transects. I conclude therefore, that results from the two techniques are directly comparable, and have no underlying biases.

Table 2B.1 Results of statistical comparison of belt and point transects. U = Mann-Whitney U statistic. P = probability. Species codes are: CX cattle, DN donkeys, ED eland, GK gerenuk, GF giraffe, GG Grant's gazelle, IM impala, OR oryx, SH shoats, TG Thompson's gazelle, WL wildebeest, ZB zebra. * = a significant difference.

Code		CX	DN	ED	GK	GF	GG	IM	OR	SH	TG	WL	ZB
N1	U	25.00	28.00	32.00	32.00	32.00	16.50	32.00	28.00	29.00	29.00	20.00	19.00
	P	0.443	0.317	1.000	1.000	1.000	0.061	1.000	0.317	0.644	0.644	0.200	0.171
N3	U	20.00	24.00	17.50	14.00	10.50	23.00	21.00	24.50	18.50	24.50	22.00	20.00
	P	0.556	0.917	0.142	0.062	0.025*	0.844	0.317	1.000	0.396	1.000	0.724	0.556
M1	U	28.00	32.00	32.00	32.00	32.00	28.00	32.00	24.50	28.00	29.00	32.00	27.00
	P	0.610	1.000	1.000	1.000	1.000	0.317	1.000	0.301	0.317	0.643	1.000	0.441
M3	U	10.00	12.50	12.50	12.50	12.50	9.00	12.50	12.50	12.50	12.50	12.00	9.00
	P	0.521	1.000	1.000	1.000	1.000	0.410	1.000	1.000	1.000	1.000	0.881	0.408
O1	U	18.00	14.00	21.00	24.50	21.00	3.50	24.50	21.00	24.00	15.00	23.00	13.00
	P	0.333	0.062	0.317	1.000	0.317	0.003*	1.000	0.317	0.917	0.157	0.845	0.139
O3	U	9.00	15.00	18.00	12.00	9.00	12.50	18.00	17.50	18.00	18.00	13.50	18.00
	P	0.059	0.317	1.000	0.140	0.059	0.370	1.000	0.902	1.000	1.000	0.391	1.000

Code: N = November, M = March, O = October, 1 = habitat one (open grassland), 2 = habitat two (thinly bushed grassland)

APPENDIX 2C
CLASSIFICATION OF MONTHS INTO SEASONS

Classification of months into seasons

Both the quantity of grass available (biomass) and its greenness were used to determine season, as an objective classification system. Any months with grass of 25% greenness or above were classified as wet season months. Below the critical 25% threshold, biomass deviations from the overall biomass mean were used to determine season. Details are explained in Chapter 2, Section 2.2.5. The results are presented in Table 2C.1 below.

Table 2C.1 Classification of months into seasons

Month	*% Green*	*Biomass: % deviation from mean*	*Season*	*Comment*
January	44.06	19.66	Wet	
February	39.18	2.16	Wet	
March	7.34	-14.95	Dry	
April	63.08	7.62	Wet	
May	57.45	57.30	Wet	
June	16.81	21.21	?	Classified as wet due to high % green & biomass
July	2.53	9.86	?	Classified as dry due to low % green & biomass
August	2.41	-3.37	Dry	
September	3.31	-25.73	Dry	
October	1.71	-56.20	drought	Later re-classified as dry season
November	36.51	-35.54	Wet	
December	47.42	18.01	Wet	

October was re-classified as a dry season month as it was the only month falling into the drought category and this would have been too small a sample size on its own.

APPENDIX 3A

MULTIPLE LOGISTIC REGRESSION ANALYSES TO INVESTIGATE FACTORS AFFECTING THE DENSITY DISTRIBUTION OF WILD GRAZERS.

Background

Chapter 3 presents results of binary logistic regression analyses which used presence/absence data to investigate factors affecting the distribution of wild grazers on Mbirikani and Merueshi Group Ranches. The results presented in this Appendix take the regression analysis one stage further, using multiple linear regressions to investigate factors affecting grazer distributions given that they were present.

Methods

Methods for collection of all the data used in these analyses are described in Chapter 3 (Section 3.2), and the variables used in the regressions are the same as those used for the binary logistic regressions in Chapter 3 (see Table 3.1). For these multiple linear regression analyses however, all transects with zero wild grazers sighted were removed in order for the assumptions of the test to be met. These analyses therefore illustrate which factors are important in affecting wild grazer density distributions, in areas where they were present.

In a similar method as for the binary logistic regressions, four scenarios were analysed independently; Mbirikani dry and wet season distributions and Merueshi dry and wet season distributions. For each scenario, potential problems of multi collinearity were addressed by examining correlations within the data (using Pearson's correlations) and collinearity diagnostics (VIF and tolerance scores, eigenvalues and variance proportions). The Durbin-Watson statistic, alongside histograms and normal probability plots were examined to ensure the models did not violate any of the assumptions of the test. An inspection of Mahalanobis distances, Cook's distances and leverage values ensured all outliers or highly influential values were noted and investigated.

As with the binary logistic regressions, the enter method was used in all cases in order to provide as much information as possible, and to remain consistent with the results presented in Chapter 3, where the enter method was found to be most suitable model selection method.

Results

In all four scenarios, the dependent variable used was the density of wild grazers log transformed.

Wet season results

Mbirikani - wet season

The Durbin-Watson test statistic (1.591) indicated the model did not violate assumptions of independent errors. The model was significant overall ($F_{139,146}$=4.451, P<0.001) and showed that both livestock density and % grass greenness had a significant positive relationship with the density of wild grazers, given that they were present. Results are given in Table 3A.1

Table 3A.1 Results of multiple linear regressions for Mbirikani wet season grazer densities (N=147)

	Unstandardized Coefficients (B) ± Std. Error	*95% CI for B*	*Standardized coefficients Beta*	*t*	*Sig.*	
(Constant)	-1.307 ± 1.481	-4.235 – 1.621		-.883	.379	
Livestock densities	.001 ± .000	.001 – .002	.274	3.418	.001	**
Distance to boma	.000 ± .000	.000 – .000	-.082	-.739	.461	
Biomass	.000 ± .001	-.001 – .002	.085	.679	.498	
%CP	.010 ± .068	-.125 – .144	.013	.141	.888	
%DOM	.034 ± .024	-.013 – .082	.136	1.420	.158	
% green	.005 ± .001	.002 – .007	.290	3.612	.000	***
Distance to water	.000 ± .000	.000 – .000	.011	.146	.884	

Adjusted R^2=0.142, Durbin-Watson=1.591, **=P<0.01, ***=P<0.001

Merueshi - wet season

Large standardised and studentized deleted residuals and Dfbeta values above 1 suggested one case was having an especially large influence on the model. Further investigation of this point showed that it was not an error, but represented an observation of a disproportionately large herd of grazers (780 individuals, with the next largest herd size being 123 and the rest below 100). Since this point was considerably non-representative of the majority of the data, and since it precluded the generation of an accurate, unbiased model, it was excluded from the analysis. The final model produced a significant result overall ($F_{59,66}$=2.509, P=0.025), and produced grass biomass as the only variable to have a significant positive relationship with wild grazer density. All results are shown in Table 3A.2.

Table 3A.2 Results of multiple linear regressions for Merueshi wet season grazer densities (N=67)

	Unstandardized Coefficients (B) ± Std. Error	*95% CI for B*	*Standardized coefficients Beta*	*t*	*Sig.*	
(Constant)	-5.193 ± 4.367	-13.930 – 3.545		-1.189	.239	
Livestock densities	.000 ± .000	-.001 – .000	-.180	-1.428	.159	
Distance to boma	.000 ± .000	.000 – .000	-.185	-.987	.328	
Biomass	.001 ± .000	.000 – .001	.332	2.538	.014	*
%CP	.019 ± .031	-.044 – .082	.079	.616	.540	
%DOM	.107 ± .074	-.042 – .255	.202	1.441	.155	
% green	.001 ± .002	-.003 – .005	.057	.437	.664	
Distance to water	.000 ± .000	.000 – .000	.055	.290	.772	

Adjusted R^2=0.138, Durbin-Watson=1.871, *=P<0.05

Dry season results

Mbirikani - dry season

Collinearity issues with %CP and biomass meant that %CP had to be removed from the analysis. There was highly significant negative relationship between biomass and %CP (r=-0.985, P<0.001). Once removed, the final model produced a significant result overall ($F_{74,80}$=4.398, P=0.001), although none of the independent variables came out as significant. All results are shown in Table 3A.3.

Table 3A.3 Results of multiple linear regressions for Mbirikani dry season grazer densities (N=81)

	Unstandardized Coefficients (B) ± Std. Error	*95% CI for B*	*Standardized coefficients Beta*	*t*	*Sig.*
(Constant)	-1.099 ± 2.256	-5.595 – 3.397		-.487	.628
Livestock densities	.000 ± .001	-.002 – .001	-.060	-.580	.563
Distance to boma	.000 ± .000	.000 – .000	-.027	-.163	.871
Biomass	-.002 ± .001	-.005 – .001	-.210	-1.440	.154
%DOM	.047 ± .039	-.030 – .124	.130	1.219	.227
% green	.017 ± .012	-.006 – .041	.153	1.493	.140
Distance to water	.000 ± .000	.000 – .000	-.231	-1.378	.172

Adjusted R^2=0.203, Durbin-Watson=1.659

Merueshi - dry season

The independent variable % green was removed from this analysis as it was consistently zero. In addition, two cases were removed from the analysis due to having anomalously high Mahalanobis distance scores and studentized residuals above 3. Both were found to have good biological reasons for exclusion as well (abnormally high livestock densities). However, even with the removal of these statistical outliers, the model did not have a good fit, and did not produce a significant result ($F_{34,40}$=0.649, P=0.698). The results are given in Table 3A.4.

Table 3A.4 Results of multiple linear regressions for Merueshi dry season grazer densities (N=41)

	Unstandardized Coefficients (B) ± Std. Error	*95% CI for B*	*Standardized coefficients Beta*	*t*	*Sig.*
(Constant)	3.396 ± 4.833	-6.425 – 13.217		.703	.487
Livestock densities	.001 ± .001	-.001 – .004	.199	1.175	.248
Distance to boma	.000 ± .000	.000 – .000	.120	.607	.548
Biomass	.001 ± .001	-.001 – .002	.172	1.025	.313
%CP	.075 ± .088	-.104 – .255	.216	.855	.399
%DOM	-.056 ± .098	-.254 – .143	-.167	-.569	.573
Distance to water	.000 ± .000	.000 – .000	.003	.018	.985

Adjusted R^2=-0.056, Durbin-Watson=2.116

Discussion

The results from these multiple linear regressions do not add much to the conclusions drawn in Chapter 3 from the binary logistic regression analyses of presence/absence data. No factors came out as significant in the dry season analyses, which may be due to the low sample sizes. However, this also suggests that other factors may be involved, and merits further research. In the wet season, the density distributions of Mbirikani wild grazers were significantly positively associated with livestock densities and grass greenness. For Merueshi grazers in the wet season, only biomass was significant, which is the same as the results investigating factors affecting presence/absence. This tentatively suggests that Mbirikani grazers may have a greater freedom of movement than Merueshi grazers, being able to positively select greener grass which is more nutritious (Sinclair 1974), and having no constraint due to livestock presence. In fact, livestock are usually taken to the best pastures to graze, so the positive association between wild grazers and livestock on Mbirikani suggests that wild grazers are also selecting the best areas.

APPENDIX 4A

HOUSEHOLD QUESTIONNAIRE – MBIRIKANI GROUP RANCH (2006)

SECTION A

1) Background: (Fill out 1.1-1.4 at the start and 1.5-1.8 at the end of the interview. No need to ask anything).

1.1 Date: .... 1.2 Interviewer: ....

1.3 Location: .... 1.4 Group Code: ....

1.5 Co-operation scale:	1	2	3	4	5	1 = very low, 2 = fairly low, 3 = average, 4 = fairly high, 5 = very high.
1.6 Knowledge scale:	1	2	3	4	5	
1.7 Attitude to wildlife scale:	1	2	3	4	5	Circle the appropriate number
1.8 Honesty scale:	1	2	3	4	5	

2) Respondent information:

2.0 Have you lived in this area for over 2 years? Yes or No ....

2.1 Are you a member of Mbirikani Group Ranch? Yes or No ....

2.2 Name (optional): .... 2.3 Sex: ....

2.4 Age set: Iltareto (1) / Ilmadidiani (2) / Ilnyangusi II (3) / Ilkololik (4) / Ilkishumu (5) / Ilkidotu (6) / Ilkiponi (7)

2.5 Clan 2.6 Subclan:

Ilaiser (1) – Ilpartimaro/Irpasingo (1) / Loodokishu (2) / Iseker (3) / Ingidongi (4)

Ilmolelian (2) – Ilmasagwa (1) / Iloiger (2) / Ilpasekero (3) / Ilmamasita (4) / Ilmakesen (5) / Ilmoingo (6) / Iltarhosero (7)

Ilaitayiok (3) – Irmamai (1) / Irmosejwa (2) / Irpojos (3) / Irkisikon/Isekei (4) / Isiria (5) / Irmoshono (6)

2.7 Marital status [married (1), single (2), widowed (3), divorced (4)]: ....

2.8 Highest level of formal education (be exact): ....

3) Household information:

3.1 Name of headperson (optional - should be your respondent): ....

3.2 Number of wives: ....

3.3 Number of unmarried children per wife: (please fill out the following table with the number of each child of each wife):

	Wife 1	Wife 2	Wife 3	Wife 4	Wife 5	Wife 6	Wife 7
Female							
Male							

3.4 Are there any other people who depend on you? (prompt them to think of nephews, brothers, sisters, mothers, grand-sons, adopted people, friends. This means anyone who depends on the household head for food).

Yes or No ....

3.5 If yes:

Number of other adult men (i.e. brothers / father) (over circumcision age): ....

Number of other adult women (i.e. sisters / mother) (over circumcision age): ....

Number of other children (including employees) (under circumcision age): ....

4) Livestock information:

4.1 Does your household own any livestock? Yes or No: .... (if no – skip to section 5, if yes – continue)

I would like to ask you about your livestock herds. To make it easier to give me numbers, I'll break it down into different types of animals: Let's start with cattle.

4.2 Cattle: (Total: ....)

Bulls	Steers	Adult female cows	Calves

4.3 Sheep: (Total: ....)

Rams	Castrated rams	Adult females	Lambs

4.4 Goats: (Total: ....)

Billy goats	Castrated billy goats	Adult females	Kids

4.5 Milk: Now I'd like to ask about your milk production from your livestock. I'm going to ask about a 'wet season' and a 'dry season'. The wet months for this study are November 2004 until June 2005. The dry months are July to October 2005, i.e. before the drought. (give them a chance to remember that time and to think about what they were doing then)

4.6 Do you produce any milk from your cows? Yes or No ....
If yes - How many litres per cow in the wet season? ....
How many litres per cow in the dry season? ....

4.7 Do you produce any milk from your goats? Yes or No ....
If yes – How many litres per goat in the wet season? ....
How many litres per goat in the dry season? ....

4.8 Do you produce any milk from your sheep? Yes or No ....
If yes – How many litres per sheep in the wet season? ....
How many litres per sheep in the wet season? ....

(If you cannot answer these questions, may we talk to your wife or child or employee who does the milking?)

Disease:

4.9 I am now interested in a few specific diseases. Can you tell me how many cattle you had to treat, specifically for Oldikana /Lipis (ECF), Enkaaya-Olokuny / iingati (MCF) and Entorobo (Nagana) during the last WET season (Nov 04 to June 05). And how many shoats you had to treat for Kurru-nkonyek (eye disease) (Write the disease in column 1, followed by the drug they used, followed by the number of bulls, cows and calves they treated for that disease, using that drug).

WET:

CATTLE	Drugs	Bulls	Steers	Cows	Calves
Oltikana					
Entorobo					
Iingati					
SHOATS	**Drugs**	**Adult Sheep**	**Lambs**	**Adult goats**	**Kids**
Kurru - Nkonyek					

4.10 Can you tell me how many cattle you had to treat, specifically for Oldikana /Lipis (ECF), Enkaaya-Olokuny / iingati (MCF) and Entorobo (Nagana) during the last DRY season (July 05 to Oct 05). And how many shoats you had to treat for Kurru-nkonyek (eye disease). (Write the disease in column 1, followed by the drug they used, followed by the number of bulls, cows and calves they treated for that disease, using that drug).

DRY:

CATTLE	Drugs	Bulls	Steers	Cows	Calves
Oltikana					
Entorobo					
Iingati					
SHOATS	**Drugs**	**Adult Sheep**	**Lambs**	**Adult goats**	**Kids**
Kurru - Nkonyek					

Death of Livestock. Introduce this topic very carefully.

(This is likely to be a sensitive topic. Explain that you are aware of this and are sorry about the death of their animals, but insist that the questions are important, as there may be ways of reducing livestock loss).

4.11 CATTLE - Did any of your cattle die during the WET season (Nov 04 to June 05)? Yes or No .... (If yes – proceed with this section. If no – go to 4.16).

4.12 How many cattle died from the following causes during the wet season (i.e. November 2004 until June 2005)? (get total number then split into bulls, steers, cows, calves)

Cause	Total cattle	Bulls	Steers	Cows	Calves
Carnivore predation (see 4.13 below)					
Hunger or thirst (drought)					
Bloat (empongit)					
Injury caused by elephants or buffalo					
• East Coast Fever (Oldikana / Lipis)					
• Malignant Cattah Fever (Enkaaya-Olokuny/Iingati)					
• Nagana (Entorobo)					
Other diseases (what?)					
Other cause (what?)					

4.13 If they had anything killed by carnivores: Where and when did the predation on your cattle happen? What carnivore was responsible?

What........................Where................................

When........................Carnivore..........................

What........................Where................................

When........................Carnivore..........................

What........................Where................................

When........................Carnivore..........................

4.14 How many calves were born in the wet season? ....

4.15 How many of these calves died before they were weaned? ....

4.16 Did any of your cattle die during the DRY season (July 2005 until October 2005)? Yes or No? ....

4.17 How many cattle died from the following causes during the dry season (July 2005 until October 2005)? (get total number then split into bulls, steers, cows, calves)

Cause	Total cattle	Bulls	Steers	Cows	Calves
Carnivore predation					
Hunger or thirst (drought)					
Bloat (empongit)					
Drinking too much water					
Injury caused by elephants or buffalo					
• East Coast Fever (Oldikana / Lipis)					
• Malignant Cattah Fever (Enkaaya-Olokuny/lingati)					
• Nagana (Entorobo)					
Other diseases (what?)					
Other cause (what?)					

4.18 If they had anything killed by carnivores: Where and when did the predation on your cattle happen? What carnivore was responsible?

What.........................Where...............................

When.........................Carnivore..........................

What.........................Where...............................

When.........................Carnivore..........................

What.........................Where...............................

When.........................Carnivore..........................

4.19 How many calves were born in the dry season? ....

4.20 How many of these calves died before they were weaned? ....

4.21 How many cattle died from drought during Jan and Feb 2006? **(DROUGHT)**

Bulls	Steers	Females	Calves

Shoat disease:

4.22 Did any of your sheep or goats die in the wet season (between November 04 and June 05)? Yes or No? ....

4.23 – IF YES: How many sheep died during the wet season of the following causes?
How many goats died during the wet season of the following causes?

Cause	**Adult sheep**	**Lambs**	**Adult goats**	**Kids**
Carnivore predation				
Hunger or thirst (drought)				
Bloat (empongit)				
Injury caused by elephants or buffalo				
Kurru-nkonyek (eye disease)				
Disease (other)				
Other cause (what?)				

4.24 If they had anything killed by carnivores: Where and when did the predation on your shoats happen? What carnivore was responsible?

What......................Where...............................

When......................Carnivore.........................

What......................Where...............................

When......................Carnivore.........................

4.25 Did any of your sheep or goats die in the dry season (July 05 to October 05)?

Yes or No? ....

4.26 – IF YES: How many sheep died during the dry season of the following causes?

How many goats died during the dry season of the following causes?

Cause	**Adult sheep**	**Lambs**	**Adult goats**	**Kids**
Carnivore predation				
Hunger or thirst (drought)				
Bloat (empongit)				
Injury caused by elephants or buffalo				
Kurru-nkonyek (eye disease)				
Disease (other)				
Other cause (what?)				

4.27 If they had anything killed by carnivores: Where and when did the predation on your shoats happen? What carnivore was responsible?

What......................Where...............................

When......................Carnivore.........................

What......................Where...............................

When......................Carnivore.........................

Tick-spraying and dipping costs:

4.28 Do you spray any of your livestock with acaricides? Yes or No? ....

4.29 If yes: Ask the following questions to fill out the table below:

- Do you use a 20L mutungi when you spray your livestock?
- What quantity of acaricide do you put into one 20L Jerry can?
- How many Jerry cans would you use per spraying session for each livestock type?
- How many times would you spray livestock per month in the wet season?
- How many times would you spray livestock per month in the dry season?

What acaricide do you use? ....

Type of Livestock	**Cattle**	**Shoats**
Quantity of acaricide per 20L Jerry can		
Number of sprays in wet season?		
Number of sprays in dry season?		
Total numbers of Jerry can per spray		

4.30 Where do you buy your acaricide? (tick all that apply – no need to rank).
(1) Mbirikani .... (2) Kimana .... (3) Emali .... (4) Loitokitok .... (5) Other ....

4.31 What diseases are you trying to prevent by spraying your livestock? (List all they mention).

....

....

Dipping:

4.32 Have you dipped your livestock in the past 2 years? Yes or No.........(if no – go to 4.39)

4.33 If yes: how often in the wet season?

....

4.34 How much does this cost per cow per month? ....

4.35 How much does this cost per shoat per month? ....

4.36 How often do you dip in the dry season? ....

4.37 How much does this cost per cow per month? ....

4.38 How much does this cost per shoat per month? ....

4.39 How many litres of deworming mixture did you use in the wet season (Nov 04 – June 05)? (Show bottles of different sizes to get best estimate – they call bottles of any size 1litre!)

....

What kind? ....

4.40 How many litres of deworming mixture did you use in the dry season (July – Oct 05)?

....

What kind? ....

4.41 Do you employ any herders for your livestock? Yes or No ....

4.42 If yes: How many? ....

4.43 If yes – why? (Rank in order of importance)

(1) To guide livestock to best pastures ....

(2) To prevent livestock getting lost in the bush ....

(3) To try to reduce carnivore depredation ....

(4) To treat and take care of livestock ....

4.44 If there was no wildlife on this ranch, would you still employ a herder for your livestock? ....

4.45 How much do you pay your herders per year?

Herder 1: ....

Herder 2: ....

Herder 3: ....

5) Agriculture (Shamba) Information:

5.1 Does your household own any shambas? Yes or No: .... (if no, skip to section 6. If yes, continue)

5.2 If YES: How many shambas does your household own? ....

5.3 What size is each shamba in acres?

Shamba 1: .... acres

Shamba 2: .... acres

Shamba 3: .... acres

6) Employment / Business Information: (Introduce this topic fully and explain the information we will be asking for – i.e. including salary information).

6.1. Do you or any member of your household have any kind of permanent paid employment?

Yes or No .... (if no – go to question 6.5)

Who? Name in Full:..

6.2. If yes – what is the job? (describe in full)

....

....

6.3 What is the wage per month? Ksh......................

6.4 Where is this job? (write all the required information on the line next to the best option)

(1) On this ranch (which area?) ....

(2) On a nearby ranch (which one?) ....

(3) In Kimana / Loitokitok / Emali / Mombassa / Nairobi? ....

(4) Other (where?) ....

Business:

6.5 Are you or any member of your household involved in any kind of business, other than your shamba? Yes or No ....

Who? Name in Full: ....

6.6 What is the business? (describe in full)...

....

6.7 Where is the business based? (write all the required information on the line next to the best option)

(1) On this ranch (which area?) ....

(2) On a nearby ranch (which one?) ....

(3) In Kimana / Loitokitok / Emali / Mombassa / Nairobi? ....

(4) Other (where?) ....

7) Wildlife Costs / Benefits information:

7.1 Do you receive any income due to sharing your land with wildlife? Yes or No .... (ask the question below even if they say no – double checking...)

7.2 Do any members of your household currently:

(Prompt with all options. Write a number – i.e. 0 if no member of the household has that, or 2 if 2 children get scholarships)

- Get any cash benefits from wildlife? .... amount per month: ....ksh
- Have a job in the tourist industry? .... wage per month: ....ksh
- Have a job as a game scout? .... wage per month: ....ksh
- Have a different job that exists due to wildlife (i.e. research assistant)? .... What job? .... wage per month: ....ksh
- Go to school on a wildlife bursary? .... value: ....ksh (i.e. how much are the school fees that the bursary covers?)
- Sell crafts to tourists? .... amount earned per month..................ksh

7.3 Apart from money benefits, have you or your household benefited from wildlife in any other way (non-financial) in the past 2 years? Yes or No .... (Even if no, prompt with the following examples, write yes or no next to each. Re-assure that the information is confidential)).

- Do you eat meat from wild animals? ....
- Do you use other wild animal products (i.e. skins, tails for fly whisk, horns, parts for ornaments)? .... If yes, what? ....
- Other (what?): ....

7.4 What problems have you had because of sharing your land with wildlife, in the last 2 years? (Rank what they say in order of importance – DO NOT prompt with the different options).

(1) Livestock predation by carnivores: ....

(2) Livestock death or injury from elephants or buffalo: ....

(3) Crop damage by wild animals: ....

(4) Disease transmission from wild animals to livestock: ....which diseases? (list all)
....

(5) Human death from wild animal: .... During protection of a shamba? ....

(6) Human injury from wild animal: .... During protection of a shamba? ....

(7) Competition for grazing / eat all the grass: ....

(8) Competition for water: ....

(9) Other: ....

(10) None: ....

7.5 IF THEY MENTIONED ANYTHING ABOUT HUMAN INJURY, WHICH WAS **NOT** RELATED TO A SHAMBA, AND **HAS** HAPPENED IN THE LAST 2 YEARS: (firstly double check that the case they are talking about was actually a member of their household and was definitely not related to a shamba / crop growing situation).

I am so sorry to hear that you have had an incident resulting in someone being hurt. I understand that this will be very hard for you to talk about, but is it possible for you to tell me what happened? (do not push the issue if they don't want to talk about it).

a) What happened? Listen to the story as told by the informant. As the relevant facts are produced, fill them in in the table below and double check at end. Make sure all the specific questions have answers- even if it means asking the person specifically):

What date did this happen? (especially the YEAR)	
How many people were injured?	
What is your relationship to these people?	
What animal was responsible?	
If the person had to receive any treatment, where were they treated (which hospital/clinic)?	
For how long was the	

person in a hospital or clinic for?	
What was the cost of the treatment?	
Who paid this?	
Was any compensation received?	
How much?	

7.6 Is it possible to get compensation if any of your livestock are killed by predators? ....

7.7 Do you or would you always report it to the Predator Compensation Program if one of your livestock is killed by predators? Yes or No ....

7.8 If no – why not?

....

....

....

8) Land ownership information:

8.1 Do you own title deeds to any land? Yes or No ....(if yes – go to 8.2, if no – go to 8.3)

IF YES:

8.2 How much land? (in acres if possible) ....

8.3 Where is this land? (full description) ....

8.4 Is the land by water? Yes or No ....

8.5 Is the land fenced? Yes or No .... 8.6 All of it or some of it? ....

8.7 Yes: (1) To prevent other peoples livestock ......

(2) To prevent wildlife getting in ....

(3) To demarcate it as my land ......

(4) Other ....

No: (5) Too expensive ....

(6) So wildlife can use it too ...

(7) So other peoples livestock can use it ....

(8) Other ....

8.8 What do you do with your land currently? (tick on the line of all the answers that apply – explain 'other' in full)

(1) Nothing ....
(2) Shamba ....
(3) Livestock farming ....
(4) Have a tourism operation ....
(5) Rent it ....
(6) Sell it ....
(7) Run a business (what): ....
(8) Other: ....

8.9 What do you plan to do with your land in the future?
(1) Nothing ....
(2) Shamba ....
(3) Livestock farming ....
(4) Have a tourism operation ....
(5) Rent it ....
(6) Sell it ....
(7) Run a business (what): ....
(8) Other: ....

8.10 Do you want to fence your land in future?......... 8.11 All of it or some of it? ....

8.12 Yes: (1) To prevent other peoples livestock ......
(2) To prevent wildlife getting in ....
(3) To demarcate it as my land ......
(4) Other ....

No: (5) Too expensive ....
(6) So wildlife can use it too ...
(7) So other peoples livestock can use it ....
(8) Other ....

IF NO (i.e. DON'T OWN ANY LAND)

8.13 Would you like to have title deeds to a piece of land? Yes or No ....

8.14 If you had title deeds to some land by water, what would you do with this land?
(1) Nothing ....
(2) Shamba ....
(3) Livestock farming ....
(4) Have a tourism operation ....

(5) Rent it ....
(6) Sell it ....
(7) Run a business (what): ....
(8) Other: ....

8.15 Would you fence this land? Yes or No ...... 8.16 All of it or some of it? ....

8.17 Yes: (1) To prevent other peoples livestock ......
(2) To prevent wildlife getting in ....
(3) To demarcate it as my land ......
(4) Other ....

No: (5) Too expensive ....
(6) So wildlife can use it too.....
(7) So other peoples livestock can use it ....
(8) Other ....

8.18 If you had title deeds to some land which was NOT by water, what would you do with this land?
(1) Nothing ....
(2) Shamba ....
(3) Livestock farming ....
(4) Have a tourism operation ....
(5) Rent it ....
(6) Sell it ....
(7) Run a business (what): ....
(8) Other: ....

8.19 Would you fence this land? Yes or No .... All of it or some of it? ....

8.20 Yes: (1) To prevent other peoples livestock ......
(2) To prevent wildlife getting in ....
(3) To demarcate it as my land ......
(4) Other ....

No: (5) Too expensive ....
(6) So wildlife can use it too.....
(7) So other peoples livestock can use it ....
(8) Other ....

Section B
(Mbirikani Group Ranch Specific)

1. Is there any land on your Group Ranch, or surrounding Group Ranches that you avoid taking your cattle to ever? Yes or No.................... (if no – go to question 4)

2. If yes: why?

(1) To give grass a chance to grow for later ....
(2) Olopololis for calves ....
(3) Other peoples land ....
(4) Protected areas (NPs)........
(5) Wildlife disease ....
(6) Other ....

3. If wildlife related - where? (be very specific)

....

....

4. If wildlife related - when and for how long?

....

5. Why do you build bomas? (rank the following in order of importance)

(1) To keep livestock warm at night ....
(2) To prevent livestock from wandering away at night and getting lost ....
(3) To protect livestock from people (stealing) ....
(4) To protect livestock from carnivores ....

6. Do you like or dislike wild HERBIVORES on your ranch? (circle the appropriate answer)

like (1) no preference (2) dislike (3)

7. Why?

Like:
(1) Bursaries for others ....
(2) Bursaries for us ....
(3) Cropping benefits to ranch ....
(4) They do not eat livestock ....
(5) Attractive to look at ....
(6) Provide useful products ....
(7) Attract tourists ....
(8) Attract projects ....
(9) Create jobs ....
(10) Other....................................

Dislike:
(11) Bring disease ....
(12) Compete for grass ....
(13) Compete for water ....
(14) Cause injury to livestock ...
(15) Cause crop damage ... .
(16) Dangerous to people ....
(17) Environmental destruction
(18) Damage bomas ....
(19) Other..........................

9) Household Income:

What percentage of your household income comes from:

a) Livestock

b) Shambas

c) Wildlife-related income (tourism employment, bursaries, game scout employment etc)

d) Other employment or business

Livestock: ....%

Shamba: ....%

Wildlife related income: ....%

Other employment / business: ....%

9.1 If wildlife were to be *more* profitable than livestock, would you be prepared to decrease your livestock herds to encourage more wildlife? Yes or No ....

9.2 If wildlife became *equally as* profitable as livestock, would you like your income to come from wildlife or livestock? (tick the answer given)

(1)Wildlife......... (2)Livestock......... (3)Either (I don't mind)......... (4)I don't understand.........

8. Do you like or dislike wild CARNIVORES on your ranch? (circle the appropriate answer)

like (1) no preference (2) dislike (3)

9. Why?

Like:
(1) Bursaries for others ....
(2) Bursaries for us ....
(3) Due to compensation project ....
(4) Create jobs ....
(5) Attractive to look at ....
(6) Provide useful products ....
(7) Attract tourists ....
(8) Attract projects ....
(9) Other ....

Dislike:
(10) Eat our livestock ....
(11) Dangerous to people ......
(12) Cause injury to livestock...
(13) Damage bomas ....
(14) Other ....

10. Do you want to GET RID OF any wildlife from your ranch? Yes or No ....

11. If yes – which species would you like to get rid of most? (If they want to get rid of ALL wildlife, and don't distinguish the most important ones – just write 'ALL' on line 1. Write a maximum of 5 in order of importance)

1) ....
2) ....
3) ....
4) ....
5) ....

12. Why?

(1) Eat our livestock ....
(2) Bring disease ....
(3) Compete for grass ....
(4) Compete for water ....
(5) Cause injury to livestock ....
(6) Cause crop damage ....
(7) Dangerous to people ....
(8) Environmental destruction ....
(9) Damage bomas ....
(10) Other...

13. Are there any species of wildlife you would like to see CONSERVED on this ranch? Yes or No ....

14. If yes – which ones, in order of importance?

1) ....
2) ....
3) ....
4) ....
5) ....

15. Why?

(1) Because of Compensation Project ....
(2) Bursaries for others ....
(3) Bursaries for us ....
(4) Cropping benefits to ranch ....
(5) They do not eat livestock ....
(6) Attractive to look at ....
(7) Provide useful products ....
(8) Attract tourists ....
(9) Attract projects ....
(10) Create jobs ....
(11) Other ....

16. Do you agree or disagree with the following statements? (Tick the relevant box)

Statement	Agree (1)	No preference (2)	Disagree (3)	Don't know (4)
Wildlife could bring lots of income to your household, if managed properly.				
Wildlife should be conserved on Mbirikani GR after land subdivision.				
An area should be set aside as a wildlife conservation area when the land subdivided				

All landowners should fence their land to keep away wildlife.				
Wildlife costs you more than what it pays you (i.e. you get more losses than benefits from wildlife).				

17. Are you for or against the subdivision of Mbirikani Group Ranch? (make sure they understand what subdivision means: dividing up the communal land and giving each member his own plot with individual title deeds)

(1) For .... (2) Against ....

18. Why? Can you give me all your reasons in order of importance. (Rank all answers that they say, but do NOT prompt with different options. Rank them in order of importance and write the number. Answer questions 19 and 20 if they said 'for' and 21 and 22 if they said against).

19. If FOR:

(1) For security of tenure / so I can own my own land ....

(2) So we can make our own decisions / manage own land ....

(3) So I can sell the land ....

(4) Because of dissatisfaction with the committee ....

(5) For equality between rich and poor / communal system benefits only the rich ....

(6) So I can rent my land or grass ....

(7) So I can subdivide between my sons ....

(8) So I can build a permanent house ....

(9) So I have access to loans ....

(10) So we can prevent other ranch members using our land ....

(11) So we don't have to have wildlife on our land ....

(12) So if people want wildlife on their land, they can have it ....

(13) So I can remain in one place and not have to move ....

(14) Other: ....

....

20. You have expressed positive reasons for subdividing the land. Despite wanting land subdivision to go ahead, are there any problems you can think of that will happen after land subdivision?

Yes or No ....

If yes: what? (Rank what they say in order of importance – do NOT prompt with different options)).

(1) I will have to decrease livestock numbers which is a problem...................
(2) There will not be enough grazing for our livestock.............
(3) Not being able to move so far will give us problems, especially in drought..............
(4) There will not be enough land for the whole family..............
(5) Wildlife will decrease.................
(6) People will sell their land...................
(7) Boundary conflicts ....
(8) Other: ....
....

21. If AGAINST: why are you against land subdivision? (Rank what they say in order of importance – do NOT prompt with different options)

(1) I will have to decrease livestock numbers, which is a problem ....
(2) There will not be enough grazing for our livestock ....
(3) Not being able to move around freely will give us problems, especially in drought ......
(4) There will not be enough land for the whole family ....
(5) Wildlife will decrease and we like wildlife ....
(6) We will lose our income from wildlife ....
(7) Others depend on our land for a grazing refuge and if we subdivide, they cant come ...
(8) It will make us poorer ....
(9) People will sell their land ....
(10) Boundary conflicts ....
(11) Other ....

22. You have said that you don't want the land to be subdivided. But is there anything you can think of that is good about land subdivision? Yes or No ....

If yes: what? (Rank what they say in order of importance – do NOT prompt with different options).

(1) For security of tenure / so I can own my own land ....
(2) So we can make our own decisions / manage own land ....
(3) So people can sell the land ....
(4) To get rid of the committee system ....

(5) It would help to create equality between rich and poor ....

(6) So people can rent land or grass ....

(7) So people can have access to loans ....

(8) Other ....

APPENDIX 4B
AN INVESTIGATION OF FACTORS AFFECTING DISEASE PREVALENCE

Introduction

During the investigation of costs facing pastoral Maasai (Chapter 4), wildlife-related diseases were found to constitute a considerable cost. In addition, costs to these diseases were found to differ significantly between regions. This appendix uses multiple linear regression analyses to investigate what variables were important in affecting the prevalence of these diseases.

Methods

Three separate multiple linear regression analyses were carried out to investigate the factors affecting the prevalence of east coast fever (ECF), malignant catarrhal fever (MCF) and trypanosomiasis. In each case the dependant variable was the estimated cost of losses to that disease, transformed where necessary to meet assumptions of normality of residuals and equality of variance. Independent variables used in the regressions are shown in Table 4B.1.

Table 4B.1 Independent variables used in regression models to identify factors influencing disease prevalence.

	Variable	***Description***
Household variables	Household size	continuous variable
	Education of household head	continuous variable - number of years of education
Livestock / Husbandry variables	Cattle herd size	continuous variable
	Density of livestock	continuous variable - the density of all livestock within the area
	Distance to nearest veterinary support	continuous variable - distance of household's boma to the nearest town with a veterinary supply store
	Cost of non-wildlife related diseases	continuous variable
	Subdivision dummy	dummy variable, 1 = subdivision and 0 = communal
Environmental variables	Woody vegetation density	continuous variable (removed for MCF regression)
	Distance to lava	continuous variable - a proxy for distance to both buffalo and tsetse fly presence
	Grass biomass	continuous variable
Wildlife variables	Wildlife density	continuous variable - the density of all wild macro-herbivores within the area (replaced with density of wildebeest in the MCF regression)

Data for the following variables were collected during the questionnaire survey; household size, education of household head, cattle herd size and cost of non-wildlife related diseases. Values for distance to nearest veterinary support and distance to lava were obtained using the 'nearest features v3.8a' extension in ArcView GIS v3.2. Woody vegetation density scores were taken from the mean density of woody vegetation recorded on vegetation transects done in the habitat in which the interview group was located. Values for density of livestock, grass biomass and wildlife density were collected during a year of monthly belt transects (see Chapter 2) and allocated to interview groups using Dirichlet tessellations.

Multiple linear regressions were carried out using the Statistical Package for Social Scientists (SPSS) v12.0. The backwards stepwise approach was used for these regressions as this consistently produced the best adjusted R^2 values. Normality of residuals, VIF and tolerance scores, colinearity diagnostics and adjusted R^2 values were all examined to ensure the final model met assumptions, had a good fit and was not biased by colinearity issues. Where Kruskal-Wallis tests indicated significant differences between groups, post-hoc testing was carried out by hand using the procedure outlined in Siegel & Castellan (1988).

Results

There was a significant difference between groups in costs from wildlife-related diseases of cattle (Kruskal-Wallis test; H_5=36.310, P<0.001). Post hoc testing showed that group 5 had significantly higher costs than groups 2, 3 and 6, and group 4 was also significantly higher than group 3. Taking the diseases separately, there was also a significant difference between groups to costs of ECF, MCF and trypanosomiasis (H_5=29.298, P<0.001; H_5=13.799, P=0.017 and H_5=58.520, P<0.001 respectively).

For ECF and MCF, there was a significant difference in costs between wet and dry seasons, with more disease occurring in the wet months (W=35401.0, P<0.001 and W=35075.5, P<0.001 respectively). For trypanosomiasis however, there was no significant seasonal difference in costs (W=32197.5, P=0.288). From the perspective of a Maasai pastoralist, however, the total loss incurred is of far greater importance than *when* it occurred and consequently further exploration of the data was conducted using total annual losses. The total cost to wildlife-related disease is considered a proxy for prevalence of that disease.

East Coast Fever

The dependant variable was the total annual cost of ECF per household, which was square root transformed to meet assumptions of normality of residuals. The model had a good fit (adjusted R^2=0.432) and produced a significant result overall ($F_{6,170}$=23.334, $P<0.001$). Five independent variables were retained in the model, and these are summarised in Table 4B.2.

Table 4B.2 Coefficient estimates and t-test statistics for variables associated with the prevalence of East Coast Fever in the multiple linear regression analysis.

	Standardized Coefficients	*t*	*Sig.*
Cattle herd size	.246	3.437	.001
Household size	.213	3.132	.002
Non-wildlife-related disease	.288	4.304	.000
Wildlife density	.425	4.060	.000
Livestock density	.163	2.173	.031
Grass biomass	-.193	-2.110	.036

Malignant Catarrhal Fever

The dependant variable was annual cost of MCF per household. No transformation was necessary, but the independent variable 'density of woody vegetation' had to be removed because of a strong correlation with density of wildebeest, which replaced density of wildlife in this analysis (r_s=0.884, $P<0.001$). The model had a good fit (adjusted R^2=0.425) and produced a significant result overall ($F_{3,173}$=44.399, $P<0.001$). Only three variables were retained in the model. These are summarised in Table 4B.3.

Table 4B.3 Coefficient estimates and t-test statistics for variables associated with the prevalence of Malignant Catarrhal Fever in the multiple linear regression analysis.

	Standardized Coefficients	*t*	*Sig.*
Cattle herd size	.229	3.281	.001
Household size	.188	2.756	.006
Non-wildlife-related disease	.431	7.029	.000

Since there was no significant difference between groups for costs to MCF in the dry season (Kruskal-Wallis Test: H=2.665, P=0.751), a second analysis was carried out on wet season data only. The same variables were found to be significant.

Trypanosomiasis

Square-root transformed annual cost to trypanosomiasis per household was used as the dependant variable in the backwards stepwise regression. The model had a good fit (adjusted R^2= 0.546) and produced a significant result overall ($F_{7,169}$=31.252, P<0.001). Seven variables were retained in the model and these are summarised in Table 4B.4.

Table 4B.4 Coefficient estimates and t-test statistics for variables associated with the prevalence of trypanosomiasis in the multiple linear regression analysis.

	Standardized Coefficients	*t*	*Sig.*
Subdivision dummy	-.216	-2.977	.003
Cattle herd size	.167	2.623	.010
Household size	.121	1.977	.050
Non-wildlife-related disease	.423	6.919	.000
Woody vegetation density	.473	5.893	.000
Density of livestock	.296	3.647	.000
Grass biomass	-.458	-6.079	.000

Discussion

East Coast Fever

ECF can be transmitted to cattle by both wildlife and other cattle. It would be expected therefore, that both density of wildlife and density of livestock in an area would contribute to the maintenance of ECF and the regression results confirm this. Both wildlife and livestock densities had a significant positive relationship with the prevalence of ECF (Table 4B.2).

Other variables which had a significant positive relationship with ECF prevalence were cattle herd size, household size and the prevalence of non-wildlife related diseases. The relationship with cattle herd size is probably due to the increased likelihood of more animals dying the larger the herd. This also explains the relationship with household size; larger households have larger cattle herds (r_s=0.462, P<0.001). The positive correlation with non-wildlife-related disease is also probably a function of herd size, but may also indicate poor husbandry.

In the regression model output, ECF was found to be negatively associated with grass biomass, which is a proxy for grass height. This seems counterintuitive for the spread of a tick-borne disease, since ticks are far more prevalent in long grass (pers. obs.). Indeed a Spearman's correlation between prevalence of ECF and grass biomass shows a significant positive relationship (r_s=0.208, P=0.005). This merits further investigation.

This analysis showed that prevalence of ECF was related in some way to household, livestock, husbandry, environmental and wildlife variables, and therefore cannot be blamed entirely on wildlife.

Malignant Catarrhal Fever

The only factors which appeared to significantly affect prevalence of MCF were those that are related to the size of the livestock holding; cattle herd size, household size and occurrence of non wildlife-related diseases. The density of wildebeest did not come out as significant, although this is unsurprising because households from all over the ranch utilise the pastures where the wildebeest calve, not just those households nearest the wildebeest concentrations. Since it is known that the prevalence of MCF is exclusively associated with young wildebeest calves (Mushi *et al.* 1981), it is not surprising that no other variables came out as significant.

Trypanosomiasis

Existing knowledge on the maintenance and transmission of trypanosomiasis indicates that wildlife presence is not a requisite factor (Grootenhuis 2000). The results of the regression analysis for trypanosomiasis (Table 4B.4) support this understanding. Whilst there is a positive correlation with prevalence of trypanosomiasis and surrounding livestock densities, the density of wildlife has no significant impact on the occurrence of trypanosomiasis in this study.

The model also shows a significant positive correlation between prevalence of trypanosomiasis and the density of woody vegetation. This is most likely due to the preference of tsetse flies for more densely wooded habitats, and because the tsetse's preferred hosts, eland and buffalo, tend to be found in densely wooded areas more so than open area (pers. obs.). The significant negative relationship with the subdivision dummy indicates that there is a higher prevalence of trypanosomiasis on Mbirikani Group Ranch (communal) as compared with Merueshi Group Ranch (subdivided). This is most likely due to the presence of more densely wooded vegetation on Mbirikani, especially lava forests, which harbour tsetse fly and buffalo.

As with ECF, there was found to be a significant positive relationship between prevalence of trypanosomiasis and both cattle herd size and household size, as well as the prevalence within the herd of non-wildlife-related diseases. These relationships are expected due to the likelihood of having more disease with bigger herds, which are usually associated with bigger households. The negative relationship between prevalence of trypanosomiasis and grass biomass needs further investigation.

Overall summary

A summary of all these results is given in Table 4B.5 to enable a more direct comparison of the factors affecting the prevalence of the different diseases.

All three diseases were significantly positively related to household size, cattle herd size and costs of non-wildlife-related diseases. This simply indicates that the bigger the household, the more cattle they have and therefore the cost of losses to disease, both wildlife-related and non-wildlife-related is likely to be higher. Whilst these values are certainly correlated, an examination of the VIF and tolerance scores in the SPSS output suggested no problem with multicollinearity, and thus no reason to exclude any variables. Additionally, Field (2006) suggests that only correlations above 0.80 should be of concern, and this was not the case here.

Table 4B.5 Summary of significance of all variables affecting prevalence of each disease (Tryp. = trypanosomiasis).

	Variable	*ECF*	*MCF*	*Tryp.*
Household variables	Household size	++	++	+
	Education of household head			
Livestock /	Cattle herd size	++	++	++
Husbandry variables	Density of livestock	+		+++
	Distance to nearest veterinary support			
	Cost of non-wildlife related diseases	+++	+++	+++
	Subdivision dummy			—
Environmental	Woody vegetation density			+++
variables	Distance to lava			
	Grass biomass	–		—
Wildlife variables	Wildlife density	+++		

+ = significant at P<0.05, ++ = significant at P<0.01 and +++ = significant at P<0.001 with a positive relationship. Negative signs indicate the same significance of negative relationships

The significant positive relationship of both ECF and MCF with the density of livestock indicates that livestock play an important role in the maintenance and spread of these diseases. Only the prevalence of ECF was significantly influenced by the surrounding densities of wildlife, suggesting that wildlife is important in the maintenance of ECF, as well as livestock. The prevalence of trypanosomiasis was the only disease related to the density of woody vegetation, for reasons given. In addition trypanosomiasis was the only disease for which prevalence differed significantly between group ranches. Both ECF and trypanosomiasis were found to be significantly negatively related to grass biomass, a result which merits further attention

The level of education of the household head, the distance to the nearest veterinary support and the distance to lava did not significantly affect the prevalence of any of the diseases. This suggests that the diseases are not related to husbandry: more educated household heads and households closer to good veterinary support are not any less likely to lose cattle to these diseases than households far from veterinary supplies and with poorly educated heads.

Conclusion

The results presented here help to justify the method used in Chapter 4 which attributes only 10% of the cost of ECF and trypanosomiasis to wildlife. The regressions show both are significantly more likely to occur at higher livestock densities and ECF is also more likely to be prevalent at higher wildlife densities. There are also other factors involved in the prevalence of both these diseases, once again suggesting wildlife cannot be entirely blamed.

APPENDIX 4C
COMPARISON OF WILDLIFE-RELATED AND NON-WILDLIFE RELATED COSTS BY GROUP

Comparison of wildlife and non-wildlife-related costs.

In Tables 4C.1 and 4C.2, wildlife-related costs are compared with non-wildlife-related costs including drought, diseases which are unrelated to wildlife, and other causes such as deaths from eating plastic bags, birthing problems and accidental deaths. Values for cattle and shoats are given independently. In addition to the mean costs in US$ per household, the percentage of the total value of the herd lost is also presented for a fair comparison between groups. An estimate of *value* lost was used rather than *numbers* of livestock, as it is a more realistic representation of actual losses: losing one calf is very different from losing one bull.

Table 4C.1 Mean costs (US$ per household per year) from cattle deaths. Values in parentheses represent the percentage of the total value of the cattle herd lost. Sig = significance of Kruskal-Wallis tests comparing the groups.

	Cattle losses Group	*1*	*2*	*3*	*4*	*5*	*6*	*Mean*	*Sig*
Wildlife related	Predation	29.3 (0.3%)	15.5 (0.2%)	12.3 (0.3%)	12.9 (0.2%)	39.6 (0.6%)	29.3 (0.5%)	23.2 (0.3%)	*
	Wildlife disease	385.1 (3.4%)	230.4 (3.4%)	143.1 (3.3%)	478.8 (7.8%)	877.1 (14.2%)	403.5 (7.3%)	419.1 (6.2%)	***
	Total wildlife-related costs	**414.4 (3.7%)**	**245.9 (3.6%)**	**155.4 (3.6%)**	**491.7 (8.0%)**	**916.7 (14.8%)**	**432.8 (7.8%)**	**442.3 (6.6%)**	***
Non-wildlife related	Drought	426.2 (3.8%)	512.9 (7.5%)	246.7 (5.7%)	594.9 (9.7%)	1028.1 (16.6%)	998.5 (18.1%)	632.9 (9.4%)	***
	Other disease	136.2 (1.2%)	144.3 (2.1%)	227.1 (5.2%)	1006.1 (16.5%)	1440.0 (23.2%)	272.9 (5.0%)	534.0 (7.9%)	***
	Other cause	53.3 (0.5%)	37.6 (0.6%)	10.0 (0.2%)	24.5 (0.4%)	17.6 (0.3%)	70.0 (1.3%)	35.4 (0.5%)	NS
	Total non-wildlife-related costs	**615.7 (5.5%)**	**694.8 (10.1%)**	**483.8 (11.2%)**	**1625.5 (26.6%)**	**2485.7 (40.1%)**	**1341.4 (24.3%)**	**1202.3 (17.9%)**	**

* = P<0.05, ** = P<0.01, *** = P<0.001, NS = non-significant

Post hoc testing was carried out to see where the differences lay. No significant differences were found for cattle predation. For wildlife-related disease, group 3 was significantly lower than groups 1, 4 and 5, and groups 5 and 6 were also significantly different. For drought, group 3 was significantly different from groups 1 and 2. For other diseases of cattle, groups 4 and 5 were significantly higher than groups 1 and 3. Overall, for total wildlife-related costs, there were significant differences between groups 1 and 3, 3 and 4, and 5 and 6. For non-wildlife-related causes, group 3 had significantly lower costs than all other groups.

Table 4C.2 Mean costs (US$ per household per year) from shoat deaths. Values in parentheses represent the percentage of the total value of the shoat herd lost. Sig = significance of Kruskal-Wallis tests comparing the groups.

	Shoat losses Group	*1*	*2*	*3*	*4*	*5*	*6*	*Mean*	*Sig*
Wildlife related	Predation	62.0 (3.5%)	83.7 (6.1%)	25.9 (3.6%)	58.7 (4.1%)	83.0 (4.8%)	59.0 (5.3%)	62.1 (4.6%)	*
	Wildlife disease	19.7 (1.1%)	5.9 (0.4%)	1.7 (0.2%)	21.8 (1.5%)	24.2 (1.4%)	19.0 (1.7%)	15.3 (1.1%)	*
	Total wildlife-related costs	**81.7 (4.6%)**	**89.6 (6.5%)**	**27.6 (3.8%)**	**80.5 (5.6%)**	**107.2 (6.2%)**	**78.0 (7.0%)**	**77.4 (5.7%)**	**
Non-wildlife related	Drought	99.0 (5.6%)	109.3 (8.0%)	45.6 (6.3%)	21.6 (1.5%)	178.3 (10.3%)	217.3 (19.4%)	112.3 (8.3%)	***
	Other disease	199.2 (11.2%)	82.4 (6.0%)	178.7 (24.7%)	445.8 (31.2%)	589.1 (34.1%)	414.5 (37.1%)	316.3 (23.3%)	***
	Other cause	37.0 (2.1%)	56.1 (4.15%)	20.0 (2.8%)	30.5 (2.1%)	40.5 (2.3%)	32.3 (2.9%)	36.2 (2.7%)	**
	Total non-wildlife related costs	**335.2 (18.8%)**	**247.8 (18.0%)**	**244.3 (33.8%)**	**497.9 (34.9%)**	**807.9 (46.7%)**	**664.1 (59.4%)**	**464.8 (34.2%)**	*

* = P<0.05, ** = P<0.01, *** = P<0.001

Post hoc testing showed that for shoat predation, group 3 suffered significantly lower costs than groups 1 and 5, but there were no significant differences between individual groups in costs to wildlife diseases of shoats. For drought, group 1 was significantly higher than groups 3 and 4, and for other diseases of shoats, there were significant differences between group 2 and groups 4, 5 and 6, and between groups 3 and 5. Only groups 2 and 6 differed significantly in costs of other causes of shoat deaths. Overall, for wildlife-related costs of shoats, group 3 was significantly lower than groups 1, 4 and 5, and for non-wildlife-related costs, only groups 3 and 5 were significantly different.

Discussion

Table 4C.1 shows that for cattle, costs from wildlife-related diseases are considerably higher than costs from predation. Total wildlife-related costs are three times lower than non-wildlife-related costs. The most important non-wildlife-related cost for most groups was drought, followed by disease. Groups 4 and 5 however lost more to disease than to drought. The major non-wildlife-related diseases of cattle were reported to be contagious bovine pleuro pneumonia (CBPP) and anthrax.

For shoats however, costs to predation were higher than costs to wildlife-related disease, which is the opposite of the cattle results. Predation losses accounted for a mean of 4.6% of the shoat herd (Table 4C.2) as compared with only 0.3% of the cattle herd. For non-wildlife-related causes of shoat deaths, in all groups except one (group 2), costs from other diseases were substantially higher than costs from drought. The main reported non-wildlife-related diseases of shoats were lumpy skin disease, contagious caprine pleuro pneumonia (CCPP) and enterotoxaemia (red-intestine disease).

For both cattle and shoats, costs from all causes of death differed significantly between groups, with the exception of 'other causes' of cattle losses. Group 5 consistently had the highest costs, both from wildlife-related and non-wildlife related causes, for both cattle and shoats, and group 3 had the lowest. Results comparing wildlife versus non-wildlife-related causes are summarised in Table 4.5.

Wildlife-related costs

Table 4C.1 shows that for cattle, costs from wildlife-related disease are considerably greater than costs from predation. The most economically significant wildlife-related disease was MCF. This is largely due to the lack of any vaccine against, or cure for MCF, resulting in almost 100% fatality. The best way to avoid the disease is to avoid grazing on pastures where wildebeest calves are present, but this is rarely possible for Maasai households who already struggle with finding sufficient grazing for their livestock. The prevalence of this disease was sufficient to create negative attitudes towards wildlife, especially wildebeest (see Chapter 5), and veterinary efforts should prioritise attempts to find a cure for this disease.

For shoats, costs to predation were higher than costs to wildlife related disease, largely because there were no economically significant shoat diseases that were spread by wildlife. Predation of shoats cost more than that of cattle, as the former appear to be easier targets for predators (Mizutani *et al.* 2005). S. Maclennan (unpublished data) found that on Mbirikani Group Ranch, more than twice as many shoats were killed per month as were cattle.

Non-wildlife-related causes

Table 4C.1 illustrates that for cattle, the most important non-wildlife-related cost was drought, followed by other diseases. For groups 4 and 5 however, non-wildlife-related diseases cost more than drought. As previously mentioned, the most economically significant non-wildlife-related diseases of cattle were contagious bovine pleuro pneumonia and anthrax. The exceptionally high cost of losses to these diseases reported by groups 4 and 5 households (on average $1006 and $1440 per year respectively) may be partially explained by the larger cattle herd sizes (under-reported here) and the habitat in which the groups were found. Unsurprisingly, groups 1, 2 and 3 which lie along the water pipeline and swamps experienced the lowest costs from drought related losses.

Table 4C.2 shows that for shoats, non-wildlife-related diseases were more costly than drought, and losses from these diseases constituted a considerable percentage of the value of the herd (23% on average). The main diseases in this category included a capripox virus related to the lumpy skin disease of cattle, enterotoxaemia (a clostridial disease commonly known as 'red-intestine disease') and contagious caprine pleuro-pneumonia (CCPP). Anthrax was also reported in shoats, especially in group 6.

APPENDIX 4D
CALCULATION OF THE COSTS OF GRAZING COMPETITION

Competition for resources by wildlife can potentially constitute a considerable cost to livestock farmers. One way of calculating the potential cost is to determine what percentage of the total grazing resource is utilised by wildlife. This calculation is illustrated below for Mbirikani Group Ranch in 2005.

Table 4D.1 Calculation of the costs of grazing competition; the density, metabiomass and grazing requirements of livestock and wildlife on Mbirikani GR. Weight = mean female body weight (used as the mean weight of all individuals in a population). TLU = Tropical Livestock Unit (body weight/250), which scales all species to a weight unit. Metabolic weight = Body weight$^{0.75}$; a reliable expression of an animals nutrient demand on the primary resource. Grass % in diet from Crawford (1968). Density = species densities (from 2005 census counts, Chapter 2). Metabiomass = the mean metabolic weight on an area (density *metabolic weight). Grass requirements = the amount of grass required by each species (metabiomass *grass %/100).

Species	*Weight (kg)*	*TLU*	*Metabolic weight*	*Grass (%)*	*Density (no/km^2)*	*Meta-biomass (kg/km^2)*	*Grass Requirement (kg/km^2)*
Livestock							
Cattle	180	*0.7*	*49.1*	*75*	26.3	1292.4	969.3
Shoats	18	0.1	8.7	70	17.2	150.3	105.2
Donkey	200	0.8	53.2	95	0.2	10.6	10.1
SUM					**43.7**	**1453.4**	**1084.6**
%					**74.3%**	**68.5%**	**68.5%**
Wildlife							
Elephant	1725	6.9	267.7	65	0.0	0.0	0.0
Giraffe	750	3.0	143.3	0	0.8	114.7	0.0
Burchell's zebra	200	0.8	53.2	98	5.0	265.9	260.6
Thomson's gazelle	15	0.1	7.6	82	0.3	2.3	1.9
Grant's gazelle	40	0.2	15.9	32	2.1	33.4	10.7
Kongoni	125	0.5	37.4	98	0.1	3.7	3.7
Impala	40	0.2	15.9	45	0.6	9.5	4.3
Wildebeest	123	0.5	36.9	100	5.6	206.8	206.8
Buffalo	450	1.8	97.7	90	0.0	0.0	0.0
Eland	340	1.4	79.2	27	0.3	23.8	6.4
Lesser kudu	70	0.3	24.2	0	0.0	0.5	0.0
Gerenuk	25	0.1	11.2	0	0.2	2.3	0.0
Oryx	150	0.6	42.9	88	0.1	4.3	3.8
SUM					**15.1**	**667.2**	**498.1**
%					**25.7%**	**31.5%**	**31.5%**

As explained in Chapter 4 however, this overall figure of 31.5% does not necessarily mean that one could replace all the wildlife on the land with 31.5% more livestock. This is due to the lack of dietary overlap whereby wildlife uses parts of the grazing resource that livestock does not. Nonetheless, there is bound to be some competition between the two, and this is discussed further in Chapter 4.

APPENDIX 5A
REASONS GIVEN FOR LIKING OR DISLIKING HERBIVORES AND CARNIVORES

Respondents were asked whether or not they liked herbivores and carnivores on their land, and the responses are given in Chapter 5, Section 5.3.3. The reasons given for their like or dislike are presented in this Appendix.

Table 5A.1 summarises for each ranch the reasons given for *liking* herbivores and carnivores.

Positive attitudes

Table 5A.1 Frequencies and percentages of responses for why respondents liked herbivores and carnivores on their ranch.

	Why like herbivores?				*Why like carnivores?*			
	Mbirikani		Merueshi		Mbirikani		Merueshi	
	fr	%	fr	%	fr	%	fr	%
Bursaries	58	26	3	20	28	18	0	0
Attract tourists & create jobs	49	22	1	7	51	32	1	50
Cropping & conservation projects	40	18	5	33	19	12	0	0
Attractive & provide useful products	26	12	3	20	6	4	1	50
Other	9	4.1	3	20	4	3	0	0
Don't eat livestock	37	17	0	0	--	--	--	--
Compensation	--	--	--	--	49	31	0	0
TOTAL	219	100	15	100	157	100	2	100

This shows that for Mbirikani, wildlife bursaries had the greatest positive influence on peoples' attitudes to herbivores, followed by the perception that herbivores attract tourists and create jobs. Despite the fact that wildlife cropping (quota-based culling of wildlife to bring income to communities) was banned in 2003, people still cite this as a reason for liking herbivores, alongside the conservation projects which bring in money through employment of research assistants. The simple fact that herbivores do not eat livestock was mentioned often as a reason for liking them. For Merueshi households, who received no financial benefits at the time of the survey, the main reason given for liking herbivores was cropping which they use to benefit from. They also mentioned wildlife bursaries because they could see their neighbours benefiting by receiving these.

As mentioned in Chapter 5, regarding carnivores, for Mbirikani households, the perception that these species attract tourists and create jobs had the greatest influence on people's attitudes, followed by the presence of the Predator Compensation Fund (PCF). Education bursaries and conservation projects were also important reasons given for liking carnivores. On Merueshi only one person claimed to like carnivores, saying that they bring in tourists and are attractive to look at.

However, grouping results by ranch, as in Table 5A.1 may mask important regional differences. Chi-square analyses with post-hoc testing indicated whether there were differences in reasons between groups. There was a significant difference in reasons given for liking herbivores (χ^2_{20}=59.82, P<0.001); see Table 5A.2. For carnivores, since only one person on Merueshi Group Ranch (group 6) claimed to like living with carnivores, this group was excluded from the chi-square analysis. The chi-square test on the remaining five groups showed a significant difference between groups in the reasons given for liking carnivores (χ^2_{12}=22.62, P=0.0312). The results are indicated in Table 5A.2.

Table 5A.2 Simplified results of the chi-square analysis of reasons for liking herbivores and carnivores by region. ns indicates that the observed frequency was not significantly different from the expected. + or - = P<0.05 and ++ or -- = P<0.01 in a positive and negative direction respectively.

	Reasons for liking herbivores						*Reasons for liking carnivores*				
Reason/Group	*1*	*2*	*3*	*4*	*5*	*6*	*1*	*2*	*3*	*4*	*5*
Bursaries	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Cropping and conservation projects	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Attract tourists and create jobs	ns	ns	++	-	ns	ns	ns	ns	ns	ns	ns
Other (inc attractive & products)	ns	ns	ns	ns	ns	+	ns	ns	++	ns	ns
Don't eat livestock	ns	--	--	ns	++	ns					
Compensation project							ns	ns	ns	ns	ns

The inter-group differences in attitudes to herbivores merit attention (Table 5A.2). For group 3 for example, an educated group close to a major town and main road, the fact that herbivores were perceived to attract tourists and create jobs was of major importance, while the fact that herbivores did not eat livestock was not very important at all. Group 4, who were fairly marginalised, and who received the least benefits from wildlife, did not perceive herbivores as important for attracting tourists and creating jobs. For group 5, who had major problems with predation (see Chapter 4), the simple fact that herbivores did not eat livestock was of great relevance. For the Merueshi respondents (group 6), who received no direct benefits, the 'other' category, which included attractiveness and use of wildlife products, was of the greatest importance.

There were few significant differences between regions on Mbirikani in reasons provided for liking carnivores. The PCF was available to everyone, and most people professing to like carnivores recognised the benefits they bring from tourism, jobs and bursaries. However Group 3 households mentioned 'other' reasons significantly more often than would be expected by chance; these included the inherent attractiveness of the carnivores and the fact that they can provide useful products (mainly trophies).

Negative attitudes

Despite the benefits from wildlife experienced by Mbirikani members, there was still a considerable proportion of household heads who reported to dislike both herbivores (36%) and carnivores (55%). The vast majority of Merueshi households disliked both; 76% disliked herbivores and 97% disliked carnivores. Reasons given are shown in Table 5A.3.

Table 5A.3 Frequencies and percentages of responses for why respondents disliked herbivores and carnivores on their ranch.

	Why dislike herbivores?				*Why dislike carnivores?*			
	Mbirikani		Merueshi		Mbirikani		Merueshi	
	fr	%	fr	%	fr	%	fr	%
Disease	43	32	17	25	--	--	--	--
Resource competition	33	24	39	57	--	--	--	--
Damage crops	21	16	2	3	--	--	--	--
Environmental destruction	12	9	2	3	--	--	--	--
Dangerous to people & livestock	17	13	3	4	77	45	28	47
Other	9	7	5	7	5	3	1	2
Eat our livestock	--	--	--	--	81	47	28	47
Damage bomas	--	--	--	--	10	6	3	5
TOTAL	135	100	68	100	173	100	60	100

The spread of disease was the major reason given by Mbirikani household heads for disliking herbivores. The most important wildlife-related disease perceived by the Maasai in this area was malignant catarrhal fever (MCF), from young wildebeest (see Chapter 4). Resource competition and crop damage were the next most important reasons. On

Merueshi however, competition for resources was the major determinant of negative attitudes to herbivores, which is consistent with their perception that resource competition was the cause of most conflict with wildlife. Disease transmission from wildlife also had an important influence on Merueshi households' attitudes. Other reasons given for the dislike of herbivores related mostly to elephants and included environmental destruction and their physical threat to people and livestock. Unsurprisingly, the main reasons given for disliking carnivores were that they killed livestock and posed an additional threat to people and livestock. They also damaged bomas.

An exploration of the differences between groups showed that there was a highly significant difference between groups in reasons given for disliking herbivores (χ^2_{15}=77.10, P<0.001). These are illustrated in Table 5A.4. However, there were found to be no significant differences between groups for reasons for disliking carnivores (χ^2_{15}=18.61, P=0.2322) so no further investigation was done.

Table 5A.4 Simplified results of the chi-square analysis of reasons for disliking herbivores by region. ns indicates that the observed frequency was not significantly different from the expected. + or - = P<0.05 and ++ or -- = P<0.01 in a positive and negative direction respectively.

Reason/Region	*1*	*2*	*3*	*4*	*5*	*6*
Bring disease	0	0	0	0	0	0
Resource competition	0	0	0	--	--	++
Damage crops, people and livestock	0	-	++	++	++	--
Other (inc boma damage & env destruction)	0	0	0	++	0	0

Table 5A.4 shows the inter-group differences in the reasons for disliking herbivores. Groups 3, 4 and 5 all mentioned that herbivores damaged crops, and could injure people and livestock more than would be expected by chance. Merueshi households (group 6) reported resource competition as a reason for disliking herbivores significantly more than would be expected by chance, while groups 4 and 5 reported this less than would be expected by chance.